4. EVALUATION OF INVENTORY ENTRIES FOR CONTAMINANTS WITH "UNKNOWN" QUANTITIES

For the time period 1984–2003, no contaminant entries were identified for which estimates of the quantities were not possible. This section was retained, however, for two reasons. First, it maintains symmetry between the HDT report (LITCO 1995) and this report. Second, this section is reserved for future discussion of unknown quantities, should they be identified at a later date and this report be revised.

References for Section 4

LITCO (Lockheed Idaho Technologies Company), 1995, *A Comprehensive Inventory of Radiological and Nonradiological Contaminants in Waste Buried in the Subsurface Disposal Area of the INEL RWMC During the Years 1952–1983*, INEL-95/0310, Rev. 1, formerly EGG-WM-10903, August 1995.

5. DATA UNCERTAINTY: SOURCES AND METHODS FOR ESTIMATING

5.1 Purpose

Two primary objectives of this task were to (a) estimate the total quantity of each contaminant disposed of in the SDA during the years 1984–1993 and project the total quantity of each contaminant that will be disposed of in the SDA during the years 1994–2003, and (b) attach uncertainty bounds to these total quantity estimates. The results are reported in Section 3.

This section explains the approach to and results of the uncertainty-estimation process that led to the upper and lower bounds on the contaminant quantities. Data uncertainties that led to corrections in best estimates, due to biases, are also discussed.

Section 5.2 provides a brief, nontechnical summary of the approach. Section 5.3 addresses the applicable requirements. Section 5.4 discusses uncertainties and biases for the recent waste (1984–1993) and how they were addressed. Section 5.5 covers the same for the projected waste (1994–2003).

5.2 Summary

Section 5 presents the statistical methods for obtaining best estimates of the contaminant quantities in recent and projected waste and the uncertainties in the best estimates. The equations that are developed allow the construction of upper and lower bounds on the activity of a contaminant in recent waste (1984–1993) and in projected waste (1994–2003).

The analysis of historical documents and data uncovered two significant upwards biases that can occur in the estimation of radioactivities in waste. They are (a) a bias in the G-M counter survey method used to assay much of the waste (recent and projected) and (b) a bias in the forecasting process used by the generators of the waste (projected only). The values of the upward biases are factors of 2 and 4, respectively. Therefore, where appropriate, the best estimates were corrected for these biases. These corrections are presented in the following sections.

In addition to the biases, there are several sources of uncertainty in the best estimate that also must be estimated in order to construct upper and lower bounds on the actual quantity. The major sources identified and estimated include error in the G-M method bias correction, error in the G-M method, error in the generator forecasting bias correction, error in the generator forecasting process for projected waste, error due to the use of scaling factors when estimating radionuclide distributions, and random error. Depending on the situation, only a subset of these uncertainties is applicable.

Section 5.5 presents the four potential cases for projected waste that are combinations of the presence or absence of the two biases. By the use of standard error propagation techniques (NCRPM 1985), the applicable uncertainties are combined to produce an overall uncertainty in the best estimate, thus allowing for construction of upper and lower bounds on the actual activity.

These two biases do not apply to estimates of the quantities of nonradiological contaminants in recent or projected waste. Bounds on these quantities were established by more straightforward methods, as described later in this section.

5.3 Requirements Concerning Uncertainty Estimates

According to the EPA's *Supplemental Guidance to RAGS: Calculating the Concentration Term* (EPA 1992), one of the most important inputs for a risk assessment is the concentrations of the contaminants. EPA (1992) recommends that an average concentration be used. It also states that, because of the uncertainty associated with estimating the true average concentration at a site, the 95% upper confidence limit (UCL) of the arithmetic mean should be used. In the absence of data necessary for estimating UCLs, a value other than the 95% UCL can be used, provided the risk assessor can document that high coverage of the true population mean occurs i.e., the value equals or exceeds the true population mean with high probability. While the guidance deals with contaminant concentrations, it can be applied equally well to contaminant quantities, which is the product of the present task.

Many sources of uncertainty are inherent in the quantification of the contaminant inventory of a waste site as complex as the SDA; some of them are quite large. It is not realistic to think that the total amount of each contaminant can be estimated statistically, especially in the absence of sampling, and that rigorous 95% confidence limits can be constructed. Therefore, the approach must be based on the second recommendation of EPA (1992). That is, a value other than the 95% UCL, but analogous to it, will be provided, with reasonable justification that it provides coverage of the true total amount with high probability.

5.4 How Uncertainties and Biases Were Addressed for the Recent Waste

5.4.1 Background

The waste buried at the SDA during the years 1984–1993 originated from several generators over various time periods and consisted of many different types. The general process from waste generation to disposal is depicted in Figure 5-1. The three boxes within the dashed oval represent the steps in the process that contribute to the uncertainty in the reported contaminant quantities in a shipment.

The step represented by the first box within the oval is the measurement of radioactive waste volumes and radionuclide activities in the shipment. The uncertainty in the estimate is due to many sources of error in this measurement process. The measurement process depends on the type of waste being shipped and the waste generator.

The second box in the uncertainty oval (Figure 5-1) pertains to the nonradioactive contaminants in the waste. Nonradiological contaminants were, at best, identified on shipping records simply as being part of a shipment to the SDA. A formal process for measuring and

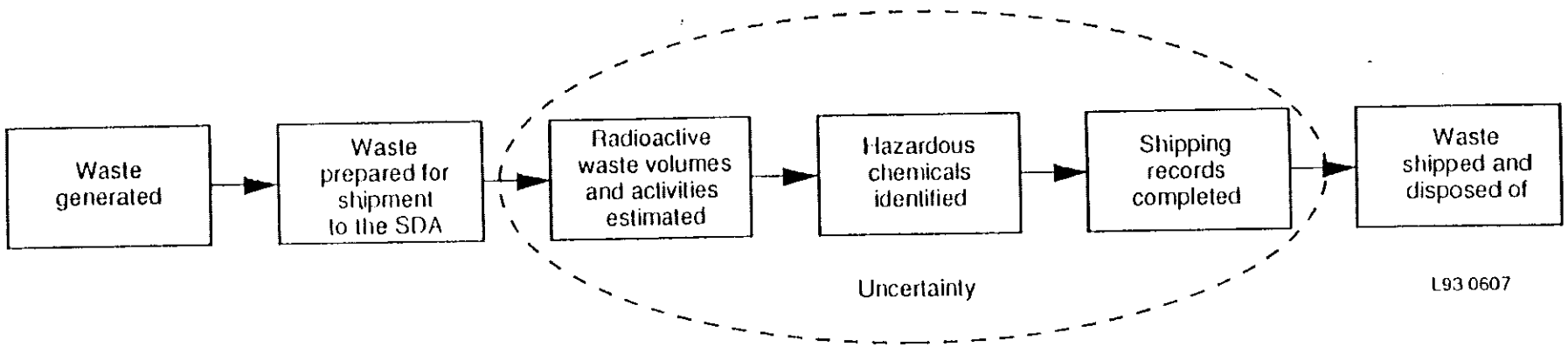

Figure 5-1. The process from waste generation to disposal.

reporting nonradiological contaminants did not exist at that time, and quantities were generally not reported on shipping records. Therefore, estimation of total quantities and uncertainties for the present task was often based on sources other than the shipping records, e.g., process knowledge and interviews with personnel acquainted with the processes that produced specific waste streams. A major source of uncertainty is the incompleteness of the available information, which tends to cause an underestimation of the total quantities.

The third box of Figure 5-1 addresses the process of recording the measurements on shipping records and transferring that information to the RWMIS database. Errors associated with transcription, summarization, interpretation, radionuclide distributions, and upper-limit reporting result in additional uncertainty in the reported total quantities of contaminants.

As discussed in Section 2, in the present task, a data form was filled out for each waste stream to record the knowledge gained in the information search. An important part of this process was the identification of major sources of uncertainty. As mentioned above, the contaminant-measurement process is dependent on the general type of waste. Furthermore, differences in the process, as well as in the magnitude of uncertainty due to each step in the process, exist among the generators. The following subsections discuss the various uncertainties.

5.4.2 Biases and Corrections for Radiological Data Originally Obtained by the G-M Counter Survey Method

For some waste streams or waste shipments, the estimate of radioactivity made at the time of shipment was based on sampling or other methods that are relatively direct in approach, or on nuclear physics calculations. However, for the majority of shipments during this period, an indirect method was used at the time of shipment to estimate the radioactivity in a container of waste. The indirect method is a major source of uncertainty in estimates of radionuclide quantities for these generators. The specific method used since the 1950s is referred to here as the G-M counter survey method or, simply, the G-M method. An additional but related source of uncertainty is the lack of specific radionuclide identification in individual waste containers. These two sources of uncertainty are discussed in this section and in Section 5.4.3 in considerable detail because of the large potential effect on the estimated radionuclide inventory.

The G-M method consists of taking radiation readings on each of the five exposed sides of a waste container using a calibrated G-M survey meter, averaging the readings, and multiplying by a constant number to convert the average radiation reading to the estimated radioactivity in curies. Several sources of uncertainty are inherent in this process. These include (a) the geometric position of the radiation source in the container, (b) the type of radiation from the particular radionuclides present in the container, (c) the density of the materials within the container (termed the "fill matrix"), and (d) the error in the survey meter itself.

Three documented studies (Simpson et al. 1982; Hartwell et al. 1987; and Hartwell and Thompson 1988) have explored the adequacy of the G-M method as applied to INEL waste containers. Although the studies involved only low-radiation-level containers, the results are believed to be generally accurate for higher-radiation-level containers, also.

The position of the source in the container appears to be a particularly large contributor to the uncertainty. According to Simpson et al. (1982), an upward bias of at least 50% (compared with more rigorous methods, such as gamma-ray spectrometry) was measured when a known MFP test source was located at the center of a mock-up waste box. (The G-M method was derived originally from theoretical considerations for steel waste dumpsters, but was applied to many kinds of waste boxes.) When the source was located away from the center of the box, biases as large as 8,500% were measured for unusual situations. The conclusion of Simpson et al. (1982) was that the G-M method is highly susceptible to overestimation of the actual curie content because of "hot spots" located near a container side and small detector-to-source distance.

Simpson et al. (1982) also noted that results using the G-M method depended on the radionuclides present in the container, compared with the radionuclides used in developing and calibrating the method. For example, if the radionuclide in the container were Co-60 and if 0.7-MeV gammas had been assigned for conversion of the radiation readings to the estimated radioactivity, the effect could be overestimation by a factor of 2 (USHEW 1970).

Another significant contributor to the uncertainty is the density of the waste container fill matrix. This contributor includes both self-shielding within the source and shielding because of other materials within the container. Hartwell et al. (1987) investigated this effect and concluded that underestimation of the actual curie content occurs even at very slight attenuation. As the fill matrix density increases, the attenuation increases, and the underestimation becomes more severe. The conversion calculation from radiation reading to curies assumes that the container offers very slight attenuation. Thus, this problem is not accounted for in the conversion. Tests conducted on various densities of fill matrix (Hartwell et al. 1987) indicated underestimation by the G-M method ranging from approximately -90% to -50% (i.e., factors of one-tenth to one-half) of the known actual value. Since the main objective was the safety of the people handling the waste, it is reasonable to assume that the fill matrix density was purposely increased when necessary to provide additional protection. Interviews have confirmed this assumption, which results in further inflation of an already significant negative bias.

Interviews with health physics personnel indicated that, during the early years, the random error in the survey meter itself was ±20%. After approximately 1976, improvements in the calibration of the meters reduced this error to ±10%.

Because of the highly variable (shipment-dependent) nature of the sources of the above uncertainty estimates, a statistically rigorous propagation to an overall uncertainty was not feasible. However, by combining professional judgment, reasonable assumptions, and standard statistical techniques, defendable bounds on actual quantities could be determined. These bounds are analogous to 95% confidence limits and represent "reasonable certainty" that they contain the true value. The following paragraphs describe the rationale used in arriving at estimates of the bias and the random error in the G-M method.

Uncertainty in the G-M method because of source position is a positive bias ranging from 50% to 8,500%, depending on the position of the source. The closer the source is to a face of the container, the more severe the bias. Typically, the contamination is not concentrated in a small volume of the container, but rather distributed throughout. A reasonable assumption is uniform distribution throughout the container. If we also assume that the bias increases

(according to the inverse of the source-to-detector distance squared) from 50% to 8,500% as the source is moved from the center of the container to a face, the resulting average bias due to source position for a uniformly distributed source is approximately 1,050%, or 11.5 times the true value.

As stated previously, the bias due to density of the fill matrix ranges from -50% to -90%, depending on the density, based on measurements of mock-up containers with known sources and fill materials ranging from air to stacked paper (specific gravity approximately 0.8) (see Hartwell et al. 1987). The majority of the actual waste containers of the time period of interest would be expected to have effective fill densities no more than that of stacked paper. (This observation is based on a review of data for waste generated more recently, and the fact that container packing density has tended to increase over the years.)

The combined bias due to source position and fill density was evaluated as follows. Based on the data described above, the largest value that could be used for the combined bias is 8,500% (a factor of 86) for source location and -50% (a factor of 0.5) for fill density, yielding a product of 4,200% (a factor of 43). The smallest value that could be used for the combined bias is 50% (a factor of 1.5) for source location and -90% (a factor of 0.1) for fill density, yielding a product of -85% (a factor of 0.15). However, these extreme values reflect highly unusual situations, such as a waste container in which a point source of radiation rests against one inner face of the container and nothing else except air is inside the container.

A more realistic set of limits on the bias was developed by assuming a uniformly distributed radiation source within the waste container. As stated above, the average bias due to source position in this case is 1,050% (a factor of 11.5). The same range of fill densities as above was retained. Thus, the largest realistic value that could be used for the combined bias is 1,050% (a factor of 11.5) for source location and -50% (a factor of 0.5) for fill density, yielding a product of 475% (a factor of 5.75). The smallest realistic value that could be used for the combined bias is 1,050% (a factor of 11.5) for source location and -90% (a factor of 0.1) for fill density, yielding a product of 15% (a factor of 1.15). A midpoint value for the combined bias is 1,050% (a factor of 11.5) for the source location and -70% (a factor of 0.3) for fill density, yielding a product of 245% (a factor of 3.45). This is the best estimate for the value of the bias. To be somewhat conservative, however, a combined bias of 100% (a factor of 2) was used for these two factors. In other words, ignoring variability because of error in the survey meter, the actual radioactivities are expected to be approximately half the value of the reported measurements using this method.

The studies by Hartwell and Thompson (1988) and Simpson et al. (1982) include the measurement of numerous actual waste containers using the more accurate gamma-ray spectrometry method and the G-M method. In all cases, the G-M method resulted in measurements exceeding those of the gamma-ray spectrometry method by percentages ranging from 10% to 3,500%. This lends some confirmation to the conservative estimate of the positive bias of a factor of 2 and to the range of realistic combined biases derived above.

While the actual energy of the radiation from the radionuclides in a waste container is definitely a contributor to error in the reported activities, it was not included here in the bias correction. This is because a large portion of the inventory is near the assumed energy level of 0.7 MeV. Radionuclides of higher energy exist in substantial quantities as well, but their effect on

the bias is to further overestimate the total quantities. To be conservative, this effect was ignored.

Thus, if the indications are that the radioactivity in a waste stream was originally estimated using the G-M method, the reported estimates of total radionuclide quantities for specific years were adjusted by dividing by two to correct for these biases and to arrive at a best estimate. This correction is an approximation because of the large numbers and varieties of waste streams and radionuclides involved. However, use of the correction is believed to result in a more accurate inventory than use of the uncorrected G-M counter readings.

The random error due to the G-M survey meter was conservatively taken to be $\pm 10\%$ for all radioactivity estimates believed to have been developed using the G-M method during the recent time period. The total random error, including the uncertainty in the bias correction itself, is developed in Section 5.4.5.

As stated previously, for certain waste streams, the data gatherers used records of direct measurements, personal knowledge, interviews, and nuclear physics calculations to arrive at a sound judgment on the uncertainty in their reported total quantities. In these cases, the data gatherers' uncertainty estimates were used to determine upper and lower bounds on the total quantities.

There are some exceptions to the approaches described above. These occurred when (a) the data gatherer lacked sufficient information to provide uncertainty estimates in the reported total quantities, and (b) the bias correction for the G-M method was not applicable. The bias correction is not applicable for radionuclides emitting weak gamma rays or no gamma rays.

CIDRA has been programmed such that, if uncertainty information does not exist in the appropriate data fields for the bounds on radionuclide quantities, it automatically calculates upper and lower bounds (see Section 5.4.5) after correcting for the G-M method bias by dividing the reported estimate by two. To ensure that these automatic calculations are not performed erroneously for radionuclides that emit very little or no gamma radiation, each waste stream was checked manually for potential occurrences of this type. Where there was any indication that the G-M method was not used for the radionuclides in question, estimates for the upper and lower bounds were provided to ensure that the G-M method correction was not applied.

Some additional considerations apply in developing the uncertainties for waste from NRF and ANL-W. These considerations are discussed in the following paragraphs.

Because high-energy-emitting Co-60 was the principal radionuclide of interest at NRF, the survey meters were typically calibrated using high-energy radiation. This adds some uncertainty in the measurement when the container holds large quantities of radionuclides emitting low-energy radiation (e.g., Fe-55 and Ni-63). These uncertainties, however, are not considered to be any more significant than other assay uncertainties. Therefore, the bias and uncertainty estimates described in this section were also applied to most of the waste from NRF. The exception was the scrap core structural material shipped from ECF in scrap casks.

In a letter dated February 27, 1989 (Bartolomucci 1989), the manager of ECF Engineering at NRF informed EG&G Idaho that ECF's past method for estimating radioactivity, or curie content, for scrap casks was in error. The letter provided revised curie content estimates, and these revised estimates were subsequently incorporated into the RWMIS database. Bartolomucci (1989) did not, however, assign uncertainty limits to the estimates.

Another letter issued by NRF (Nieslanik 1994) applied an accuracy of +10% and -30% to scrap cask activity calculations, taking into consideration incomplete content data on some cores when received, approximations that deleted radionuclides contributing less than 1% to the total activity, and assumptions that had to be made regarding radioactive flux and core life.

The method used by NRF to arrive at radioactivity estimates for the scrap cask shipments was based on knowledge of the metal alloys in the reactor core structural materials and of the reactor core radiation history. This information allowed NRF to calculate the extent of expected neutron activation of the core structural material. This activity was then decayed for the length of time from the end of reactor operation until the scrap was shipped from ECF to the SDA.

In summary, NRF's uncertainty estimate of +10% and -30% for the scrap cask estimates was used in this report, whereas the bias and uncertainty estimates in this section related to the G-M method were applied to all the other waste from NRF.

Radioactivity estimates of ANL-W waste generated after 1970 were made at the time of shipment using a refined G-M counter method. The method factored in the type of waste container and other information. This method is considered more reliable than the typical G-M counter method, which was used by all generators listed previously. Therefore, upon the advice of ANL-W technical personnel, no bias correction was applied to ANL-W waste activity measurements made beginning in 1971. The random error was specified by ANL-W personnel to be ±25% for such measurements.

For all generators, in the CIDRA database, the radionuclide quantities (including the effects of the G-M correction, if any) are listed as the "best estimates." The uncorrected quantities are also available from CIDRA and are called the "reported estimates."

5.4.3 Scaling Factor Uncertainties for Radiological Data

Another significant source of uncertainty is that due to the use of scaling factors for estimating radionuclide distributions. In fact, based on the following analysis, it appears to be the dominant source of uncertainty in estimates of the total activity of a specific radionuclide.

A scaling factor is a fraction or percentage of the activity of another radionuclide or of the total activity of a group of radionuclides. Scaling factors were used to estimate the activities of several difficult-to-measure radionuclides in waste shipments to the SDA. For example, suppose the total activity in a waste shipment is 100 Ci and the scaling factor for Sr-90 (whose activity is difficult to measure outside a laboratory) is 0.15 (15%). Then the estimated activity of Sr-90 in the shipment is 15 Ci.

Development of scaling factors was performed by evaluating the data from analytical laboratories possessing the capabilities to analyze the activities of these difficult-to-measure radionuclides and relate the activities to those of easily analyzed radionuclides or total sample activities.

The uncertainty in the scaling factor must be estimated and incorporated into the overall uncertainty in the radionuclide activity. The following paragraphs provide an overview of the development of the uncertainty estimates for the scaling factors. Einerson and Smith (1995) provides the details. Section 5.4.5 incorporates the scaling factor uncertainty into the overall uncertainty.

Limited INEL data exist on scaling factors for the waste disposed of in the SDA. The most comprehensive data available elsewhere exist in a report prepared for EPRI (EPRI 1987). That report provides the results of an extensive data collection and analysis effort, including activities of several radionuclides from various waste types and reactor types. The data most closely resembling SDA waste came from samples originating in waste from pressurized water reactors of commercial nuclear utilities.

Two basic approaches are possible for estimating the uncertainty due to the use of scaling factors. The first is to identify all of the sources of uncertainty inherent in the process of developing and using scaling factors (e.g. analytical error, error due to the G-M survey method). These uncertainties would then be propagated to obtain an overall uncertainty estimate due to the use of scaling factors. The second method is strictly empirical. This approach would involve the use of a large data set (such as that found in the EPRI report) containing the activities of several radionuclides for several waste streams. Then, by constructing scaling factors and estimating the distributional properties, the uncertainty is empirically developed.

Since a large data set that is somewhat representative of the SDA waste streams exists in the EPRI (1987) report, the empirical approach was used here. The three basic steps were to (a) choose subsets of the EPRI radionuclides thought to best represent the radionuclides present in the SDA waste, (b) estimate the scaling factor mean, standard deviation, and relative standard deviation (RSD, the uncertainty) for each radionuclide in this subset, and (c) apply these uncertainty estimates to appropriate subsets of the radionuclides and waste streams for the SDA waste. A subset of radionuclides from the EPRI data was selected because the analysis of every radionuclide would have added only minimal information.

The subset of radionuclides analyzed from the EPRI data included C-14, Fe-55, Ni-63, Sr-90, Tc-99, I-129, Co-60, and Cs-137. These radionuclides were selected because they are representative of the difficult-to-measure radionuclides present in the SDA waste and the radionuclides to which their activities are compared. Therefore, they should demonstrate the range of scaling factor uncertainties inherent in the radionuclides present in the SDA.

The scaling factor for a radionuclide is the ratio of the activity for the radionuclide to the total activity in the waste. The total activity in a sample is defined here to be the sum of the eight radionuclides given above and is shown in equation (5-1). It is recognized that, in actuality, several more radionuclides may constitute the total set. However, it seems reasonable to assume that the estimate of scaling factor uncertainty will not depend on the number of radionuclides

used when calculating a "total" activity, as long as the set of radionuclides used is representative and fairly comprehensive.

The total activity in a sample is defined here to be the sum of the eight radionuclides indicated and is shown in equation (5-1):

$$t_j = \Sigma_i a_{ij} \quad (5\text{-}1)$$

where

t_j = total activity for sample j, and

a_{ij} = activity of radionuclide i for sample j.

Then for each sample and each radionuclide used in this analysis, a scaling factor can be written as

$$w_{ij} = a_{ij}/t_j \quad (5\text{-}2)$$

where

w_{ij} = scaling factor for radionuclide i and sample j.

The uncertainty referred to above is in terms of the RSD, which is defined as the standard deviation divided by the mean. Therefore, the next step in the analysis was to estimate the mean, standard deviation, and RSD of the scaling factors for each radionuclide across all samples for waste from pressurized water reactors in EPRI (1987). The results are presented in Table 5-1, along with the number of samples comprising the estimates.

Table 5-1. Scaling factor relative standard deviations for EPRI (1987) data.

Ratio	Number of samples	RSD
C-14/total	273	3.4
Fe-55/total	268	0.9
Ni-63/total	280	1.0
Sr-90/total	234	4.8
Tc-99/total	30	4.4
I-129/total	20	3.7
Co-60/total	333	0.7
Cs-137/total	241	1.1

Logical groupings of RSD values are apparent from the results in Table 5-1. The scaling factor RSDs for Fe-55, Ni-63, Co-60, and Cs-137 are 0.9, 1.0, 0.7, and 1.1, respectively. The scaling factor RSDs for C-14, Sr-90, and Tc-99 were 3.4, 4.8, and 4.4, respectively. Based on these results, two scaling factor RSDs, 1 and 5, were chosen for application to the uncertainty estimation for the radionuclides in the SDA waste that involved the use of scaling factors. These values of 1 and 5 were chosen based on simplicity and conservatism. While it would have been possible in theory to estimate a separate RSD for each radionuclide, the effort was not warranted considering the limited additional accuracy obtainable and the limited data available.

As described in Einerson and Smith (1995), the uncertainty in the scaling factors also depends on the particular stream in which the radionuclide exists because the method of estimating the activity of a given radionuclide sometimes varied from stream to stream. Thus, the radionuclides in the SDA waste can be placed into three groups corresponding to the three possibilities of scaling factor uncertainty: RSDs of 0, 1, and 5. An RSD of zero occurs for those radionuclides for which scaling factors were not used in determining their activity.

Table 5-2 presents the scaling factor uncertainty used for each of the radionuclides when incorporating this uncertainty into the overall uncertainty of the total activities. Details of the rules for applying scaling factor uncertainties, as well as some exceptions to Table 5-2 based on the method by which the distribution was estimated for each waste stream, are presented in Einerson and Smith (1995).

Unless excluded by either or both considerations related to an RSD of 0 or an excluded waste stream, the scaling factor uncertainty was added to the other identified uncertainties, whether or not the data gatherer had listed upper and lower bounds for the radioactivity entry on the datasheets.

5.4.4 Uncertainties for Nonradiological Contaminants

For nonradiological contaminants, the main source of uncertainty is the lack of information. For some waste streams, the data gatherers obtained good estimates, with associated uncertainties, of the total quantities of particular contaminants. In these instances, the data gatherers' estimates

Table 5-2. Scaling factor relative standard deviations for use in the Recent and Projected Data Task uncertainty estimation.

Radionuclides	Scaling factor RSD used in uncertainty estimation
U, Th, Ra (all isotopes of)	0
Cs-137, Co-60, Fe-55, Ni-63	1
All other radionuclides	5

were used. These estimates are for a variety of contaminants from several waste streams and can be considered a representative subset of all the nonradiological contaminants identified. In the HDT report, the upper bounds estimated by the data gatherers ranged from 1 to 3.6 times the estimated amount, with the majority being less than a factor of 2. When lacking uncertainty information, a factor of 2, based on the data gatherer's professional judgment, was conservatively used to construct an upper bound on the total quantities disposed of. The same assumption was used for the reporting period 1984–1993.

5.4.5 Best Estimates and Bounds for the Recent Waste

Each waste stream from each waste generator was identified, and annual quantities of radiological and nonradiological contaminants in the streams were estimated. In addition to these estimates of annual quantities disposed of, bounds on these estimates were calculated. While it was not possible to calculate 95% confidence limits in the standard way because of the lack of sampling and appropriate data, it was possible to arrive at reasonable and defensible bounds based on the historical information acquired and on knowledge of the sources of uncertainty described in the preceding sections.

When possible, the bounds provided represent the data gatherers' indication that, with reasonable certainty, the true annual quantities buried are contained within them. In some cases, the data gatherers' indications are based on the particular waste stream and the measurement methods used at the time. In other cases, heavier reliance was placed on professional judgment. When professional judgment could not be made, generic error bounds were constructed by propagation of known biases and uncertainties. "Reasonable certainty" can be considered analogous to 95% confidence; while not statistically rigorous, it represents a legitimate attempt at quantifying a very difficult parameter.

With the assumption that the bounds estimated by the data gatherers (or through propagation) represent 95% confidence limits, the following discussion presents the method used to propagate the uncertainties so that uncertainty bounds could be constructed on the total amount of a contaminant disposed of at the SDA in all waste streams.

An individual contaminant may occur in a variety of forms and in a variety of waste streams. Therefore, it may or may not be useful to group all occurrences together when estimating contaminant quantities for use in a risk assessment. Groupings of contaminant occurrences will have to be performed based on the particular objectives of the data used in the risk assessment.

Once a risk assessor determines a desired grouping, all occurrences in CIDRA for which the contaminant meets the grouping specification (e.g., a particular physical form of the contaminant) are flagged. An occurrence is a single row of Part C or Part D of the data form (see Appendix A). Each row corresponds to information for one contaminant from a single waste stream for a single year (or a range of years during which the generation rate was assumed constant). A single data form is restricted to describing only a single waste stream.

After the contaminants of interest have been selected, grouped, and flagged in the database, the next step is to estimate the quantities needed by the risk assessor. These include the best

estimate of the total amount disposed of and its upper bound (analogous to a 95% UCL) for each uniquely flagged contaminant grouping.

The best estimate for the total amount of a contaminant grouping is simply the sum over all waste streams and all years for that contaminant grouping, as expressed by the following equation:

$$T = \Sigma_i \Sigma_j T_{ij} \quad (5\text{-}3)$$

where

T = best estimate of the total quantity of a particular contaminant grouping disposed of

T_{ij} = best estimate of the quantity of the particular contaminant grouping disposed of from waste stream i in year j.

To construct an upper bound on T requires s_{ij}, the standard deviations of T_{ij}. In cases where analysis data or professional judgment have been used to estimate U_{ij}, the upper bound on T_{ij}, the standard deviation of T_{ij} can be estimated as given in Equation (5-4).

$$s_{ij} = (U_{ij} - T_{ij})/2, \text{ when based on analysis data or professional judgment.} \quad (5\text{-}4)$$

When such information is not available, s_{ij} is estimated based on the biases and random error involved. For radiological contaminants, the bias was shown earlier to range from a factor of 1.15 to a factor of 5.75. Thus, a bias correction (division by the bias) would range from 0.87 to 0.17 with a midpoint of 0.5, which is the correction factor used. It is assumed that this range is an approximate 95% confidence interval on the true bias. Given this assumption, an estimate of the uncertainty s_k (one standard deviation) in the bias correction is shown in Equation (5-5).

$$s_k = \frac{\text{range of 95\% confidence interval}}{4} = \frac{0.87 - 0.17}{4} = 0.17 \quad (5\text{-}5)$$

The estimate of the uncertainty s_X due to random error in the G-M survey meter is 10% of the reported quantity, as given in Equation (5-6).

$s_{X_{ij}} = 0.1\ X_{ij}$, where $_{ij}$ is the reported quantity of a particular contaminant grouping disposed of from waste stream i in year j. (5-6)

The estimate of the uncertainty due to the scaling factor, in terms of the RSD s_w/w, depends on the specific radionuclide and waste stream, as mentioned in Section 5.4.3 and discussed in detail in Einerson and Smith (1995). The three distinct cases are RSDs of 0, 1, and 5.

Combining these uncertainties, using the method of statistical differentials (Kotz and Johnson 1988), leads to a formula for estimating the standard deviation of T_{ij}, as shown in Equations (5-7) and (5-8).

$T_{ij} = kX_{ij}$, where k is the bias correction, whose value is 0.5. (5-7)

$$s_{ij} = \sqrt{(kX_{ij})^2 \left[\left(\frac{s_k}{k}\right)^2 + \left(\frac{s_{X_{ij}}}{X_{ij}}\right)^2 + \left(\frac{s_w}{w}\right)^2 \right]} \quad (5\text{-}8)$$

$$= T_{ij} \sqrt{0.12 + \left(\frac{s_w}{w}\right)^2}, \text{ when analysis data or professional judgement are not available.}$$

For nonradiological contaminants, a conservative estimate of half the reported quantity, based on the discussion in Section 5.4.4, is used for s_{ij} when professional judgement cannot be made.

$s_{ij} = 0.5T_{ij}$, for nonradiological contaminants when professional judgement cannot be made. (5-9)

The standard deviation s of T can then be calculated as

$$s = (\Sigma_i \Sigma_j s_{ij}^2)^{1/2} \quad . \quad (5\text{-}10)$$

Data of this type typically follow a lognormal distribution (Gilbert 1987). Therefore, it is reasonable to assume that the total activity T of a radionuclide (or total quantity of a nonradiological contaminant) is lognormally distributed with mean α and standard deviation β, where α and β are estimated by T and s. Due to the relationship between the normal and lognormal distributions (Blackwood 1992), it follows that the natural logarithm of T is normally distributed with mean μ and standard deviation σ with

$$\alpha = e^{\mu + \frac{1}{2}\sigma^2} \quad (5\text{-}11)$$

$$\beta^2 = e^{2\mu+\sigma^2}\left(e^{\sigma^2}-1\right) \quad . \quad (5\text{-}12)$$

Solving for μ and σ and using T and s as estimates of α and β gives:

$$\mu = \ln(T) - \frac{1}{2}\sigma^2 \quad (5\text{-}13)$$

$$\sigma^2 = \ln\left(\frac{T^2 + s^2}{T^2}\right) \quad . \quad (5\text{-}14)$$

An upper bound on the total quantity for a particular contaminant grouping U can now be calculated as shown in Equation (5-15).

$$U = e^{(\mu + 2\sigma)} \quad . \tag{5-15}$$

The construction of a lower bound L on T is analogous to the upper bound and is given in Equation (5-16).

$$L = e^{(\mu - 2\sigma)} \quad . \tag{5-16}$$

The above approach cannot be considered statistically rigorous. However, with the combination of professional judgment, reasonable assumptions, and conservative approximations, there is reasonable certainty (i.e., 95% confidence) that the upper bounds derived with this approach are not exceeded.

5.5 How Uncertainties and Biases Were Addressed for the Projected Waste

Except for one factor, the uncertainties and biases for the projected waste were addressed by the same method as for the recent waste discussed in Section 5.4. That factor relates to the uncertainties and biases of the process by which the quantities of contaminants in future waste were estimated. The method for addressing this factor is discussed later in this section.

5.5.1 Background

The quantities of radiological contaminants in the projected waste (1994–2003) were estimated based largely on the forecasts of waste generators. (See Section 2.4 for a discussion of the method used to project contaminants in future waste.) The generator forecasts contain biases and uncertainties. To obtain best estimates of any future waste activities that were projected using the generator forecasts, the forecasts must be corrected for the biases. To obtain uncertainties in the best estimates, uncertainties in the bias corrections must be estimated and combined with other sources of error. This section provides the methods and formulae for estimating the biases and uncertainties. Additional details on the error propagation techniques used here can be found in *A Handbook of Radioactivity Measurements Procedures* (NCRPM 1985).

As discussed in Section 5.4, it was estimated that there also existed a G-M survey method bias of 100% (factor of 2 upward) requiring correction by multiplying the reported value by 0.5. It was also estimated that this bias correction has an uncertainty of 0.17 (one standard deviation). These conclusions are summarized in equations (5-17) and (5-18).

$$k = 0.5 = \text{G-M survey method bias correction} \tag{5-17}$$

s_k = 0.17 = the standard deviation of the bias correction for the G-M survey method. (5-18)

The G-M survey method will continue to be used by waste generators for assaying some of the projected waste. Therefore, the same bias correction and uncertainty due to the G-M survey method will be applied where appropriate.

Uncertainty due to the use of scaling factors was also discussed in Section 5.4 and applies to the projected waste as well. The estimated uncertainty due to scaling factors, expressed as RSDs, is 0, 1, or 5, depending on the radionuclide and how its activity was estimated in a given stream.

5.5.2 Bias and Uncertainty of Waste Generator Forecasts

The bias and uncertainty of the generator forecasts were estimated by comparing past forecasted radioactivity against the "actual"[a] disposed radioactivity reported later in RWMIS. The accuracy of the forecasts has improved slightly from their initial use in 1977 for waste to be generated in 1978. Therefore, only data from the most recent 4 years were used.

Table 5-3 summarizes data comparisons for forecasts made during 1989 for the years 1990 through 1993 (Welch 1990), during 1990 for the years 1991 through 1993 (Welch 1991), during 1991 for the years 1992 and 1993 (Welch 1992), and during 1992 for 1993 (Hutchison 1993).

The data are not 100% complete. For NRF, only generator forecasts made in 1990 and 1991 are available. For TRA, the disposed value in 1993 is due almost totally to the unanticipated disposal of beryllium reflector blocks containing 293,000 Ci. Since this particular disposal was not part of the forecast (except for the forecast made in 1992), the activity was subtracted from the reported amount before the comparisons were made (except for the reported amount corresponding to the forecast made in 1992).

For each year, the forecasted amounts and actual amounts were summed across facilities. The last three columns of Table 5-3 present these totals, along with the ratio of the forecasted total to the actual total. In every case, this ratio is greater than 1. This result indicates that a positive bias exists in the forecasting process, i.e., the generator forecasts are higher than the actual amount. Furthermore, the ratios range from 2.27 to 5.80 with a mean of 4.0 and a standard deviation of 1.11. This result indicates an average forecast bias of 300% (a factor of 4 upward). Therefore, multiplication by 0.25 is required to correct the best estimates of radioactivity in projected waste for this bias in the waste generator forecasts.

In addition to the generator forecasting bias as a source of error, there exists variability in the bias itself, as apparent from the range of ratios, and random error in the forecasting process. These two sources of variability cannot be separately estimated because there are not multiple

a. In the current discussion, the term "actual" disposed radioactivity means the radioactivity reported after the future years have come to pass and the waste has actually been generated and disposed of. The term is not intended to mean that the reported radioactivity is an errorless measurement of the radioactivity disposed of.

Table 5-3. Comparisons of forecasted and reported actual radioactivity (Ci) in waste disposed of in the Subsurface Disposal Area, for recent years and facilities where data were available.[a]

Year forecast made	Year forecast for	ANL		ICPP		D&D		TAN		TRA		WERF		NRF		Total		
		Forecast	Actual	Forecast	Actual	Forecast	Actual	Forecast	Actual	Forecast	Actual	Forecast	Actual	Forecast	Actual	Forecast	Actual	Ratio
1989	1990	276,000	130,511	360	211	1.05	<1	6,630	2,658	19,926	68	<1[b]	6	NA[c]	74,087	302,918	133,455[d]	2.27
1989	1991	280,000	82,261	360	24	<1	<1	<1	<1	3,766	2,430	<1	3	NA	102,849	284,129	84,720[d]	3.35
1989	1992	310,000	92,896	360	2	<1	<1	<1	347	3,766	909	<1	NA[c]	NA	49,795	314,129	94,155[d]	3.34
1989	1993	330,000	87,058	320	<1	<1	2	<1	216	9,166	5,152[e]	<1	<1	NA	42,259	339,489	92,430[d]	3.67
1990	1991	360,000	82,261	360	24	0	<1	<1	<1	23,500	2,430	1.32	3	136,000	102,849	519,862	187,569	2.77
1990	1992	450,000	92,896	360	2	0	<1	<1	347	3,710	909	1.43	NA	153,000	49,795	607,072	143,950	4.22
1990	1993	520,000	87,058	320	<1	0	2	<1	216	9,100	5,152[e]	1.43	<1	161,000	42,259	690,422	134,689	5.13
1991	1992	560,000	92,896	360	2	110	<1	<1	347	30,700	909	0	NA	123,000	49,795	714,171	143,950	4.96
1991	1993	448,000	87,058	320	<1	120	2	<1	216	4,620	5,152[e]	<1	<1	144,000	42,259	597,062	134,689	4.43
1992	1993	767,000	87,058	160	<1	3.76	2	<1	216	1,470,000	298,152[f]	20	<1	NA	42,259	2,237,184	385,430[d]	5.80

a. If both forecasted and actual data for at least 1 year at a facility are not available, the facility is not presented here. Also, if all data for a facility are less than 1 curie, the facility is not presented here.

b. The value of 1 was used for "<1" entries when summing for totals.

c. Data were not available.

d. These totals do not include NRF actuals since NRF forecasts were not available.

e. The reported actual had 293,000 curies removed since it was due to the unanticipated disposal of beryllium reflector blocks and therefore not part of the forecast.

f. It was assumed that the beryllium reflector blocks were accounted for in the 1992 forecasting of the 1993 amount and therefore the 293,000 curies were included in the actual.

forecasts in single years. However, the standard deviation of the ratios of 1.11 given above contains both the bias variability and random error in the forecasting process.

An estimate of the uncertainty in the generator forecast bias correction is derived as follows:

$$\bar{r} = 4.0 = \text{mean of the forecast-to-actual ratios} \tag{5-19}$$

$$a = \frac{1}{\bar{r}} = 0.25 = \text{forecast bias correction} \tag{5-20}$$

$$s_r = 1.11 = \text{the standard deviation of the forecast-to-actual ratios} \tag{5-21}$$

$$s_{\bar{r}} = \left[\frac{s_r^2}{10}\right]^{\frac{1}{2}} = \left[\frac{(1.11)^2}{10}\right]^{\frac{1}{2}} = 0.35 = \text{ the standard error of the mean ratio,} \tag{5-22}$$

where 10 is the number of ratios used to estimate s_r .

$$s_a = \left[\frac{s_{\bar{r}}^2}{\bar{r}^4}\right]^{\frac{1}{2}} = \left[\frac{(0.35)^2}{256}\right]^{\frac{1}{2}} = 0.022 = \text{the standard deviation of the forecast bias correction.} \tag{5-23}$$

In some cases, the data gatherers did not use the generator forecasts. Some other source of information was used in projecting the activity in future waste. There may be some bias in projections of this type that were made by the data gatherers. However, no projected-versus-actual data are available for this situation. Therefore, no bias was ascribed to such forecasts and no bias correction was applied. This approach is probably conservative due to historic overestimation of curies by generator forecasters.

5.5.3 Best Estimates and Bounds for the Projected Waste—General

In an analogous expression to Equation (5-3) for the recent waste, the best estimate for the total activity of a radionuclide in the projected waste is the sum of the best estimates over all waste streams and all years for that radionuclide, as expressed by the following equation:

$$T = \Sigma_i \, \Sigma_j \, T_{ij} \tag{5-24}$$

where

T = best estimate of the total activity of a radionuclide

T_{ij} = best estimate of the activity of the radionuclide from waste stream i in year j.

Upper and lower bounds, analogous to 95% confidence bounds, for the best estimate were discussed in Section 5.4.5 [Equations (5-11) through (5-16)] and are summarized in the following equations:

$$U = e^{(\mu + 2\sigma)} \tag{5-25}$$

$$L = e^{(\mu - 2\sigma)} \tag{5-26}$$

where

$$\mu = \ln(T) - \frac{1}{2}s^2$$

$$\sigma^2 = \ln\left(\frac{T^2 + s^2}{T^2}\right) .$$

As shown in the above equations, the bounds on T require s_{ij}, an estimate of the standard deviation of T_{ij}. The construction of the best estimates and bounds depends on the sources of uncertainty involved. Four cases need to be considered for the projected waste. The first is the case in which neither the G-M method bias correction nor the generator forecasting bias correction is applicable. The second is when only the G-M method bias correction is applicable. The third is when only the generator forecast bias correction is applicable, and the fourth is when both bias corrections are necessary. The following sections provide the details for obtaining best estimates and associated uncertainties for each of these four cases.

It may not be readily apparent how projections of the radioactivity in future waste can be affected by G-M counter measurements, which can be made only on waste that exists. The reason for the interaction is that many generator forecasts of activity are made by some method of extrapolating past data, with or without adjustments to reflect changes in planned operations. If the past data were obtained using the G-M method and, therefore, are biased, then the extrapolated data for future waste are similarly biased. This bias needs to be corrected to use the forecasts in the current study.

5.5.4 Best Estimates and Uncertainty for Projected Waste when Bias Corrections are Not Applicable

In the first case, the estimates of radioactivity in projected waste are based on information about recent waste for which the G-M survey method was not used. For example, laboratory results for the radioactivity in recent waste may have been extrapolated to project the radioactivity in future waste. In such cases, the G-M correction is not applicable.

Additionally, in this case, the projections are based on information obtained by the data gatherer and not on the generator forecasts. Therefore, it is assumed that the systematic over-estimation in generator forecasts, described in Section 5.5.2, does not apply.

Three sources of uncertainty still exist, however. The first is random error in the measurement method used, s_X. This uncertainty is assumed to be the same as that estimated for the G-M method (i.e. 10% of the measured value). The second is uncertainty due to the use of scaling factors, expressed as relative standard deviation s_w/w. This uncertainty is assumed to be the same as the s_w/w estimated for the recent waste. The third uncertainty is random error in the projections made by the data gatherers. Due to the lack of any data to estimate this error, the standard deviation of the forecast-to-actual ratios, s_r, will be used. As stated in the preceding section, s_r contains both variability in the forecast bias and random error in the forecasting process. Thus, it should be somewhat conservative as an estimate of the random error in the data gatherers' predictions. The following equations were used for this first case involving projected waste.

The best estimate of the activity of a radionuclide is given by

$$T_{ij} = X_{ij} \tag{5-27}$$

where

X_{ij} = projected activity of a radionuclide for waste stream i and year j.

The standard deviation of the best estimate is given by

$$s_{ij} = \sqrt{T_{ij}^2 \left[\left(\frac{s_{X_{ij}}}{X_{ij}} \right)^2 + \left(\frac{s_w}{w} \right)^2 \right] + s_r^2}$$

$$= \sqrt{T_{ij}^2 \left[0.01 + \left(\frac{s_w}{w} \right)^2 \right] + 1.23} \tag{5-28}$$

where

$s_{X_{ij}}$ = $0.10X_{ij}$ = standard deviation due to random error in the measurement method (assumed to equal that for the G-M method)

s_r = 1.11 = the standard deviation of the forecast-to-actual ratios

$\frac{s_w}{w}$ = scaling factor RSD of 0, 1, or 5, depending on the radionuclide and waste stream.

5.5.5 Best Estimates and Uncertainty for Projected Waste when G-M Method Bias Correction is Applicable but Generator Forecast Bias Correction is Not

The second case is when the estimates of radioactivity in projected waste are based on data for recent waste that were obtained using the G-M method. However, in this case, the data gatherer did not base the projection on generator forecasts, but rather on some other source of information. It is assumed that the generator forecasting bias discussed earlier does not apply. Therefore, only the G-M method bias correction is necessary. Uncertainties due to the G-M method bias correction, random error in the G-M measurement method, scaling factor uncertainty, and random error in the data gatherers' projection must be considered in estimating the overall uncertainty. As in the preceding section, the standard deviation of the forecast-to-actual ratios, s_r, will be used as a conservative estimate of the random error in the data gatherers' projections. The following equations were used for this case involving projected waste.

The best estimate of the activity of a radionuclide is given by

$$T_{ij} = kX_{ij} = 0.5X_{ij} \tag{5-29}$$

where

X_{ij} = projected activity of a radionuclide for waste stream i and year j

k = 0.5 = G-M method bias correction.

The standard deviation of the best estimate is given by

$$s_{ij} = \sqrt{T_{ij}^2 \left[\left(\frac{s_{X_{ij}}}{x} \right)^2 + \left(\frac{s_w}{w} \right)^2 \right] + s_r^2} = \sqrt{T_{ij}^2 \left[0.01 + \left(\frac{s_w}{w} \right)^2 \right] + 1.23} \tag{5-30}$$

where

$s_{X_{ij}}$ = $0.10X_{ij}$ = standard deviation due to random error in the G-M method

$\frac{s_w}{w}$ = scaling factor RSD of 0, 1, or 5 depending on the radionuclide and waste stream

s_r = 1.11 = the standard deviation of the forecast-to-actual ratios.

5.5.6 Best Estimates and Uncertainty for Projected Waste when G-M Method Bias Correction is Not Applicable but Generator Forecast Bias Correction is

In the third case, the estimates of radioactivity in projected waste are based on extrapolations of data for recent waste that were obtained without the G-M survey method. Generator forecasts are the basis for the projections and, therefore, the generator forecast bias correction is applicable. Uncertainties due to the generator forecast bias correction, random error in the measurement method, scaling factor uncertainty, and random error in the generator forecasting process must also be factored into the overall uncertainty. The following equations were used for this case involving projected waste.

From Equation (5-20), the best estimate of the activity of a radionuclide is given by

$$T_{ij} = aX_{ij} = 0.25X_{ij} \tag{5-31}$$

where

X_{ij} = forecasted activity of a radionuclide for waste stream i and year j

a = 0.25 = forecast bias correction.

The standard deviation of the best estimate is given by

$$s_{ij} = \sqrt{T_{ij}^2 \left[\left(\frac{s_a}{a} \right)^2 + \left(\frac{s_{X_{ij}}}{X_{ij}} \right)^2 + \left(\frac{s_w}{w} \right)^2 \right] + s_r^2} \tag{5-32}$$

$$= \sqrt{T_{ij}^2 \left[0.018 + \left(\frac{s_w}{w} \right)^2 \right] + 1.23}$$

where

s_a = 0.022 = standard deviation of forecast bias correction

s_X = $0.10X_{ij}$ = standard deviation due to random error in the G-M method

$\frac{s_w}{w}$ = scaling factor RSD of 0, 1, or 5, depending on the radionuclide and waste stream

s_r = 1.11 = the standard deviation of the forecast-to-actual ratios.

5.5.7 Best Estimates and Uncertainty for Projected Waste when G-M Method Bias Correction and Generator Forecast Bias Correction are Applicable

In the final case, the estimates of radioactivity in projected waste are based on generator forecasts that used data on recent waste as the basis for the forecasts. Additionally, the G-M survey method was used in measuring the activity given in that recent waste data. For this case, all sources of error described in the preceding sections are applicable when estimating the overall uncertainty. The following equations were used for this case involving projected waste.

The best estimate of the activity of a radionuclide is given by

$$T_{ij} = akX_{ij} = 0.125X_{ij} \tag{5-33}$$

where

X_{ij} = forecasted activity of a radionuclide for waste stream i and year j

a = 0.25 = forecast bias correction

k = 0.5 = G-M method bias correction.

The standard deviation of the best estimate is given by

$$s_{ij} = \sqrt{T_{ij}^2 \left[\left(\frac{s_a}{a} \right)^2 + \left(\frac{s_k}{k} \right)^2 + \left(\frac{s_{X_{ij}}}{X_{ij}} \right)^2 + \left(\frac{s_w}{w} \right)^2 \right] + s_r^2} \tag{5-34}$$

$$= \sqrt{T_{ij}^2 \left[0.133 + \left(\frac{s_w}{w} \right)^2 \right] + 1.23}$$

where

s_a = 0.022 = standard deviation of forecast bias correction

s_k = 0.17 = standard deviation of G-M method bias correction

$s_{X_{ij}}$ = $0.10X_{ij}$ = standard deviation due to random error in the G-M method

$\frac{s_w}{w}$ = scaling factor RSD of 0, 1, or 5, depending on the radionuclide and waste stream

s_r = 1.11 = the standard deviation of the forecast-to-actual ratios.

5.5.8 Uncertainties for Nonradiological Contaminants

The approach for addressing uncertainties for nonradiological contaminants in the projected waste was identical to that used in the HDT and for the recent waste. The waste generators forecasts did not address nonradiological contaminants, so the bias correction for such forecasts did not apply.

5.6 Summary of Equations for Standard Deviations

For convenience, Table 5-4 summarizes the equations used for calculating s_{ij} when laboratory analysis data and informed professional judgement are not available for a particular radionuclide entry on a data sheet. If laboratory data or informed professional judgement is available, data from that source are used, instead, for s_{ij}.

Table 5-4. Formulae for calculating standard deviation s_{ij} for radiological contaminants when laboratory analysis data and informed professional judgement are not available.

Time period	Historical	Recent	Projected
No GM correction No forecast correction	NA[a]	NA	$s_{ij} = \sqrt{X_{ij}^2 \left[0.01 + \left(\frac{s_w}{w}\right)^2\right] + 1.23}$
GM correction No forecast correction	$s_{ij} = 0.5X_{ij} \sqrt{0.16 + \left(\frac{s_w}{w}\right)^2}$	$s_{ij} = 0.5X_{ij} \sqrt{0.12 + \left(\frac{s_w}{w}\right)^2}$	$s_{ij} = \sqrt{(0.5X_{ij})^2 \left[0.01 + \left(\frac{s_w}{w}\right)^2\right] + 1.23}$
No GM correction Forecast correction	NA	NA	$s_{ij} = \sqrt{(0.25X_{ij})^2 \left[0.018 + \left(\frac{s_w}{w}\right)^2\right] + 1.23}$
GM correction Forecast correction	NA	NA	$s_{ij} = \sqrt{(0.125X_{ij})^2 \left[0.133 + \left(\frac{s_w}{w}\right)^2\right] + 1.23}$

Definition: X_{ij} = reported quantity.

a. Not applicable.

b. $\frac{s_w}{w}$ = scaling factor RSD of 0, 1, 5, depending on radionuclide and waste stream.

References for Section 5

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6. CONFIRMING THE COMPLETENESS OF THE RESULTS

This section compares the contaminant inventory against estimates given in previous reports and in existing databases, to the extent that such comparisons are possible and meaningful. In some cases, adjustments were necessary to compare values on the same basis. The inventory is also compared against the list of contaminants detected in environmental monitoring conducted at the RWMC. The results of all these comparisons help to confirm the credibility and substantial completeness of the inventory compiled in this task.

Although estimates of waste volume are included in CIDRA, no similar comparisons have been performed to confirm the accuracy of the volume estimates. The BRA will not use the volume estimates from CIDRA, so no special confirmation was considered necessary.

6.1 Comparison of Inventory with Estimates Given in Earlier Reports

Many earlier reports (see the references cited in Sections 2 and 3, for example) provide useful information on the inventories of contaminants buried in the SDA. The earlier reports were examined as part of the data gathering in the present task. However, the inventories in the earlier reports either (a) contain estimates for only a portion of the total inventory (e.g., only one disposal unit), (b) provide mostly or solely qualitative information, (c) deal with a somewhat different time period, or (d) were developed for a different purpose and made different assumptions to deal with the lack of definitive data in the original records. Therefore, only limited comparisons were possible between the total inventory developed in the present task and the inventories in previous reports. Nevertheless, even the limited comparisons are useful to help confirm the credibility and substantial completeness of the current results.

6.1.1 Nonradiological Contaminants

Background. As discussed in Section 1.2, the acceptance of mixed waste for disposal at the SDA was generally discontinued in April 1984 (Nelson 1984). The number and quantities of nonradiological contaminants in the waste being disposed of decreased dramatically after that time because most (but not all) of those nonradiological contaminants constitute hazardous waste, per RCRA.

The one temporary exception to the termination of mixed waste receipts related to the disposal of lead used as shielding in waste containers. Lead used as shielding continued to be accepted after April 1984. In March 1985, acceptance of radioactive waste containing any lead was suspended until the issue could be evaluated (Rodgers 1985). In June 1986, it was decided (Rodgers 1986) that all lead, regardless of category or use, would be barred from disposal at the SDA on November 30, 1987. On December 16, 1987, a one-time extension of the disposal deadline for lead shielding until December 31, 1987, was announced (Rodgers 1987).

Because of the various announcements just discussed, the quantities of RCRA nonradiological contaminants other than lead in waste disposed of after April 1984 would be expected to be (and were found to be) very small. No appreciable quantity of lead should have

been disposed of after 1987, and none was identified in the present study. In addition, the quantities of non-RCRA nonradiological contaminants in waste disposed of after April 1984 decreased substantially.

Comparisons. A search of INEL reports and letters identified very few studies on the nonradiological contaminants buried in the SDA during the recent period (1984 through 1993). Two documents, Cerven (1987) and Wells (1986), provided limited information on which to base some comparisons; these are discussed below.

An INEL engineering design file (Cerven 1987) addressed nonradiological contaminants in the waste buried in the SDA. The report reviewed the RWMIS data through October 30, 1987, and also included responses to a request for information on quantities of halogenated solvents disposed of at the SDA. Most of the information in Cerven (1987) relates to the waste disposed of from 1954 through 1981. Very limited information was provided on the waste disposed of from 1982 through October 30, 1987, a time period that partially overlaps the time period covered in the present study.

The RWMIS rollups from 1982 to October 30, 1987, in Cerven (1987) provided minimal information on the disposal of some lead and some asbestos. The rollups also showed some used ion-exchange resin that could have contained some unknown hazardous chemicals. The responses to Cerven's request for information revealed that generally no halogenated solvents were disposed of at the SDA after 1983. Minor (residual) amounts of halogenated solvents may have been disposed of in rags used for wiping up small amounts of commercial products such as "WD-40" or "Tap Magic."

The rollups presented by Cerven (1987) covered time periods both before and after the January 1, 1984, start date for the RPDT waste. As a result, no direct comparisons of the Cerven data against CIDRA data could be made in the present document.

A letter by Wells (1986) covers disposals during the period from November 1980 through January 15, 1986. Again, Wells (1986) covers time periods both before and after the start date of January 1, 1984, for coverage in the RPDT. Nevertheless, the letter is useful for comparison purposes.

Wells (1986) includes estimates of the amounts of lead disposed of in 1984, 1985, and early January 1986 (see Table 6-1). Table 6-1 shows that 8.0E+07 g of lead was disposed of in 1984, 9.9E+06 g in 1985, and 1.2E+05 g in early January 1986.

The quantities in Table 6-1 agree with those in CIDRA for these generators and these time periods. The agreement was to be expected because Wells (1986) was one of the information sources used in the RPDT. CIDRA includes substantial lead disposals in 1986 after January 15 and in 1987; these disposals came after the issuance of the Wells letter. Therefore, the CIDRA total for lead is somewhat larger than that in Wells (1986).

Wells (1986) provides very limited information on zirconium oxide, insufficient for direct comparisons.

Table 6-1. Lead disposal in the Subsurface Disposal Area from 1984 through January 15, 1986, taken from Wells (1986).

Description	Generator	Year	Number of shipments	Lead quantity, g (lbs)
HFEF-5 insert with lead shielding	ANL-W	1984	23	2.8E+06 (6.16E+03)
Fuel charging casks with lead shielding	ICPP	1984	4	6.3E+07 (1.38E+05)
Wooden boxes with lead	ICPP	1984	2	1.4E+07 (3.02E+04)
HFEF-5 insert with lead shielding	ANL-W	1985	16	2.0E+06 (4.29E+03)
Resin cask with lead shielding	TAN (LOFT)	1985 (5-31-85)	1	5.4E+06 (1.20E+04)
Concrete resin module with lead shielding	PBF	1985 (5-31-85)	1	2.5E+06 (5.60E+03)
HFEF-5 insert with lead shielding	ANL-W	1986 (1-8-86)	1	1.2E+05 (2.68E+02)

In conclusion, Wells (1986) and Cerven (1987) provided some useful information for confirming the completeness of the CIDRA quantities of lead in the recent period. They provided very limited information on asbestos and zirconium oxide, insufficient for direct comparisons. No reports were found for comparison with the CIDRA quantities of beryllium, copper, chromium, mercury, cadmium, or arsenic.

No documents provided comprehensive information on nonradiological contaminants in the projected waste, against which to compare the CIDRA results.

6.1.2 Radiological Contaminants

A search was made for reports containing radionuclide inventories against which to compare the CIDRA data. For valid comparisons of the CIDRA data with radionuclide inventories in other reports, several aspects of the inventories must match. These aspects include the time period under consideration, the sources of the waste, the type of waste considered, and in which part of the SDA the waste was buried. This study examined all waste buried at the SDA from all generators from 1984–1993 and all waste projected to be buried from 1994–2003.

6.1.2.1 Recent Period (1984–1993). The report by Plansky and Hoiland (1992) contains data for waste disposed of in the SDA during part of the recent period (1984–1989). The data are nearly identical to those in RWMIS. A detailed comparison was not carried out because a comparison against RWMIS for the years 1984–1993 is made in Section 6.2.

No reports were identified containing radionuclide inventories independent of those in RWMIS. Thus, no comparisons were possible against other reports containing radiological inventories for the SDA for the period 1984–1993.

6.1.2.2 Projected Period (1994–2003).

Comparisons—The waste in the projected time period has not yet been generated. Therefore, the only type of inventory document against which to compare the CIDRA results for the projected time period is the waste generator forecasts. These forecasts were one of the sources of information for CIDRA.

As discussed in Section 2.4.2, the waste forecasts from all generators are compiled in an annual letter (i.e., Randall 1994). The letter gives the total radioactivity by generator and by year, although the specific radionuclides are not indicated. No nonradiological contaminants are included in Randall (1994).

As a completeness check, the CIDRA inventory of radiological contaminants in the projected waste, by generator, was compared against the corresponding inventory based directly on the waste generator forecasts (Randall 1994). The results are given in Table 6-2.

The first column of Table 6-2 lists the six major generators. The next column lists the projected total radioactivity from each major generator, as taken from Randall (1994). The next column provides the CIDRA "reported" estimate of radioactivity (i.e., the estimates before applying the corrections for biases in the generator forecasts and in the G-M method). The last column provides the CIDRA best estimate (i.e., the estimate after applying both bias corrections, where applicable).

Evaluation of Differences for Reported Estimates in CIDRA—The reported estimates in CIDRA are virtually identical to the values from the waste generator forecasts. The totals match exactly, to within the precision of the standard two significant digits. The values for most of the individual generators match exactly or almost exactly. Because, as stated in Section 2.4.4, the generator forecasts were the starting point for evaluating the projected waste in the current study, the close agreement is not surprising.

The differences between the generator forecasts and the CIDRA reported estimates are discussed below.

The small difference of about 3 Ci for waste from TAN is due to including in CIDRA some TAN hot cell and hot shop waste that is not included in the generator forecast.

Table 6-2. Comparison of CIDRA radionuclide inventory for the projected period (1994–2003) against direct compilation of waste generator forecasts for the same years (Randall 1994).

Generator	Compilation of waste generator forecasts (Ci)	CIDRA reported estimate (no generator forecast corrections and no G-M corrections) (Ci)	CIDRA best estimate (with generator forecast corrections and G-M corrections) (Ci)
TAN	1.0E-01	3.2E+00	1.5E+00
TRA	2.8E+06	2.8E+06	2.8E+06
ICPP	1.3E+03	1.2E+03	1.6E+02
NRF	5.5E+05	5.5E+05	1.4E+05
ANL-W	3.2E+06	3.2E+06	8.1E+05
Other	2.6E+02	1.5E+03	5.1E+02
Total	6.6E+06	6.6E+06	3.8E+06

The small difference in projected radioactivity from ICPP is strictly due to rounding. If the calculations are carried out more precisely, both the detailed waste generator forecast letter from ICPP (Hitz and Skinner 1993) and the CIDRA waste datasheet for stream CPP-ALL-1P give a total radioactivity of 1,255 Ci.

For the generators termed "other," the reported CIDRA value is considerably higher than that in the generator forecasts: 1,500 Ci versus 260 Ci. The difference is attributed to higher estimates for (a) D&D waste—890 Ci in CIDRA versus 230 Ci in the generator forecasts, and (b) waste to be treated at WERF and disposed of in the SDA—510 Ci in CIDRA versus 0 Ci in the generator forecasts.

The difference in the radioactivity of waste expected to come from D&D is explained as follows. The D&D generator forecast in Randall (1994) shows no radioactivity in the D&D waste to be produced after 1998, even though increasing volumes of waste are projected after 1998. The reason for this anomaly is that the radioactivity in the D&D waste to be produced after 1998 is not well known; facility characterizations have not yet been completed. In the present study, the radioactivity in all of the D&D waste through 2003 was estimated. The method for estimating the radioactivity after 1998 was to (a) assume that the concentration of radioactivity (Ci/m^3) in 1999–2003 waste would equal that found in the 1984–1993 waste, then (b) multiply by the projected volume of the D&D waste after 1998. This approach leads to a larger value of the projected radioactivity in D&D waste in 1994–2003 than that given in Randall (1994).

The difference in the radioactivity of waste expected to come from WERF to the SDA for disposal is explained as follows. The generator forecast values in Table 6-2 apply only to waste expected to be shipped directly from the generator to the SDA. At the time when Randall (1994) was prepared, WERF was shut down. WERF personnel were not certain when, if ever, WERF would restart. Therefore, the entries in Randall (1994) showing future treated waste going from WERF to the SDA were all zero for 1994–2003. Randall (1994) also does not include the waste stored at WERF awaiting treatment, since it had already been generated in the past. Randall (1994) does, however, provide separate rollups (not included in the total) for the waste expected to go first to WERF for treatment and then to the SDA. Those separate rollups total 607 Ci. CIDRA includes the waste to be treated that is already stored at WERF. CIDRA also reflects the assumption that WERF would restart in 1994; Section 2.5.6 indicates that size reduction and compaction have restarted, and incineration is now expected to restart in 1995. All of these entries in CIDRA total 510 Ci.

Evaluation of Differences for Best Estimates in CIDRA—The best-estimate projections in CIDRA reflect the application of corrections where applicable for (a) the upward bias in waste generator forecasts, which were generally the starting point for the CIDRA projections, and (b) the upward bias in projected values of radioactivity that are based on past measurements of radioactivity using the G-M method. The first bias correction is multiplication by a factor of 0.25. The second bias correction is multiplication by a factor of 0.5. The bases for these bias corrections and the explanations of the applicability of each are provided in Section 5.

The last column of Table 6-2 shows that applying the two downward corrections where appropriate reduces the projected radioactivity, as expected. The total radioactivity is reduced by 42%, although the amount of the reduction varies from generator to generator.

In terms of radioactivity, the largest waste generator in this time period is ANL-W, with a reported estimate of 3.2 million Ci. More than 99% of the ANL-W radioactivity is in stream ANL-785-1P, remote-handled, nonprocessible, subassembly waste. As stated in Section 5, the bias correction for the G-M method does not apply to ANL-W in this time period. The bias correction for the waste generator forecasts does apply, however. Therefore, one would expect the best estimate radioactivity for ANL-W to be approximately one-fourth of the reported radioactivity. In Table 6-2, the ratio of these two quantities is 8.1E+05/3.2E+06 = 0.25, as expected.

The second largest generator is TRA, with a reported estimate of 2.8 million Ci. Approximately 98% of the TRA radioactivity is in stream TRA-670-1P, beryllium reflectors. No corrections apply to this stream for the following reasons. The G-M method was not used or extrapolated to estimate the radioactivity, which is nearly all H-3 (a pure beta-emitter not detectable by the G-M method). The waste generator forecast correction does not apply because much of the subject radioactivity has already been generated in the reflectors. Disposal of this waste stream is essentially certain to occur, although the exact year of future disposal is uncertain. Therefore, the best estimate value of the projected radioactivity from TRA matches the reported value.

The third largest generator is NRF, with a reported estimate of 550,000 Ci. More than 99% of the NRF radioactivity is in stream NRF-618-8P, structural components removed from Navy

nuclear fuel modules. For this stream, the G-M correction does not apply because core operating histories and nuclear physics calculations were used, as discussed in Section 5. The bias correction for the waste generator forecasts does apply, however. Therefore, one would expect the best-estimate radioactivity for NRF to be approximately one-fourth of the reported radioactivity. In Table 6-2, the ratio of these two quantities is 1.4E+05/5.5E+05 = 0.25, as expected.

The three remaining major generators are projected to generate much smaller activities than the three just evaluated. For each generator, if both corrections applied to all streams, the best-estimate radioactivity would be one-eighth of the reported radioactivity. If neither correction applied to any streams, the two values would be identical. For TAN, ICPP, and other generators, the ratios are 0.47, 0.13, and 0.34, respectively. These values all fall within the expected range of 0.12 to 1.00.

Conclusion—For the projected time period, the CIDRA values of radioactivity agree with the waste generator forecasts in the instances where such agreement would be expected. The CIDRA values differ from the generator forecasts in the expected way when the CIDRA bias corrections and other refinements are applied. There is no indication that any substantial radioactivity is missing from the CIDRA data for the projected time period.

6.2 Comparison of Inventory with Inventories in Existing Databases

6.2.1 Introduction

This section compares the contaminant inventory developed in this task with corresponding inventories in existing databases. One objective was to confirm the substantial completeness and accuracy of the data collection for this task. A second objective was to identify and explain any major differences in inventory values between the databases, and justify the new values that will be used in the BRA.

Only one database was identified against which to compare the complete contaminant inventory: RWMIS. Since RWMIS contains little information on nonradiological contaminants in the waste and no estimates of uncertainties, the comparisons here involved only reported estimates and best estimates of radiological contaminants. Also, because waste in the projected time period has not been generated and the data have not been entered into RWMIS, only the recent period is addressed.

Because of the thousands of data involved in the radionuclide inventory, the comparisons reported here were for the purpose of general checking. The comparisons were not intended to be an exact accounting (which would not be useful, anyway, because of the uncertainties in the data).

6.2.2 Comparisons at the Level of Individual Radionuclides, Summed Over All Generators

6.2.2.1 Approach. The RWMIS shipping record rollups were used for these comparisons against CIDRA. Figure 6-1 illustrates the approach. The strategy was to check for agreement first at the level of the total inventory of each radionuclide (over all waste generators). If, for a given radionuclide, the numbers were not reasonably close at that level, resolution was sought by comparisons at the level of the individual waste generators. Because CIDRA is organized by waste stream and RWMIS is organized by waste shipment, direct comparisons below the generator level were generally not feasible.

As the upper-right portion of Figure 6-1 shows, before the activities could be compared on the same basis, the RWMIS results had to be adjusted to replace the generic terms MAP, MFP, and unidentified beta-gamma with specific estimates by radionuclide. (Approximately 7,000 Ci of the RWMIS radioactivity for 1984–1993 is listed in these generic terms.) The radionuclide distributions used in CIDRA for MAP, MFP, etc., vary by waste generator and sometimes even by waste stream for the same generator. *For purposes of this comparison only*, approximate breakdowns were developed as follows for each of the generic terms in RWMIS. For each generator, radionuclide distributions were identified that had been used in CIDRA, either for all waste streams or as a rough average (see Appendix E for the detailed distributions). These percentages were then multiplied by the RWMIS value, in curies, for each generic term for each generator. The resulting activities of each radionuclide were then added to the RWMIS values for the specific radionuclides. For example, the Co-60 activities deriving from the MAP value and from the unidentified beta-gamma value were added to the Co-60 activity that was listed separately in RWMIS. This process was performed for each affected radionuclide for each generator.

There is an additional complication. Section 5.4 noted that the radioactivity determinations for most waste containers from most facilities were based on radiation surveys using G-M counters. The bias and random error of that method were discussed. A correction factor—multiplication by 0.5—was derived. CIDRA applies that correction factor to all best-estimate inventory entries for which uncertainties were not available, except as discussed in Section 5. Unfortunately, applying the correction factor makes it difficult to compare RWMIS and CIDRA as a completeness confirmation for CIDRA. For ease of comparison, the initial comparisons were made without the factor of 0.5 incorporated. The final comparisons reflect the factor of 0.5 where appropriate, as shown at the bottom of Figure 6-1.

6.2.2.2 Inventories as Listed in RWMIS and CIDRA. This section discusses how the inventory information was assembled for the comparisons. The successive columns of Table 6-3 indicate the results at various stages of the comparisons.

The first two columns of Table 6-3 list the total inventory for each radionuclide, as given in the RWMIS shipping record rollups. The radionuclides are listed in order of activity. The activities listed for the generic terms MFP, MAP, etc., are evident.

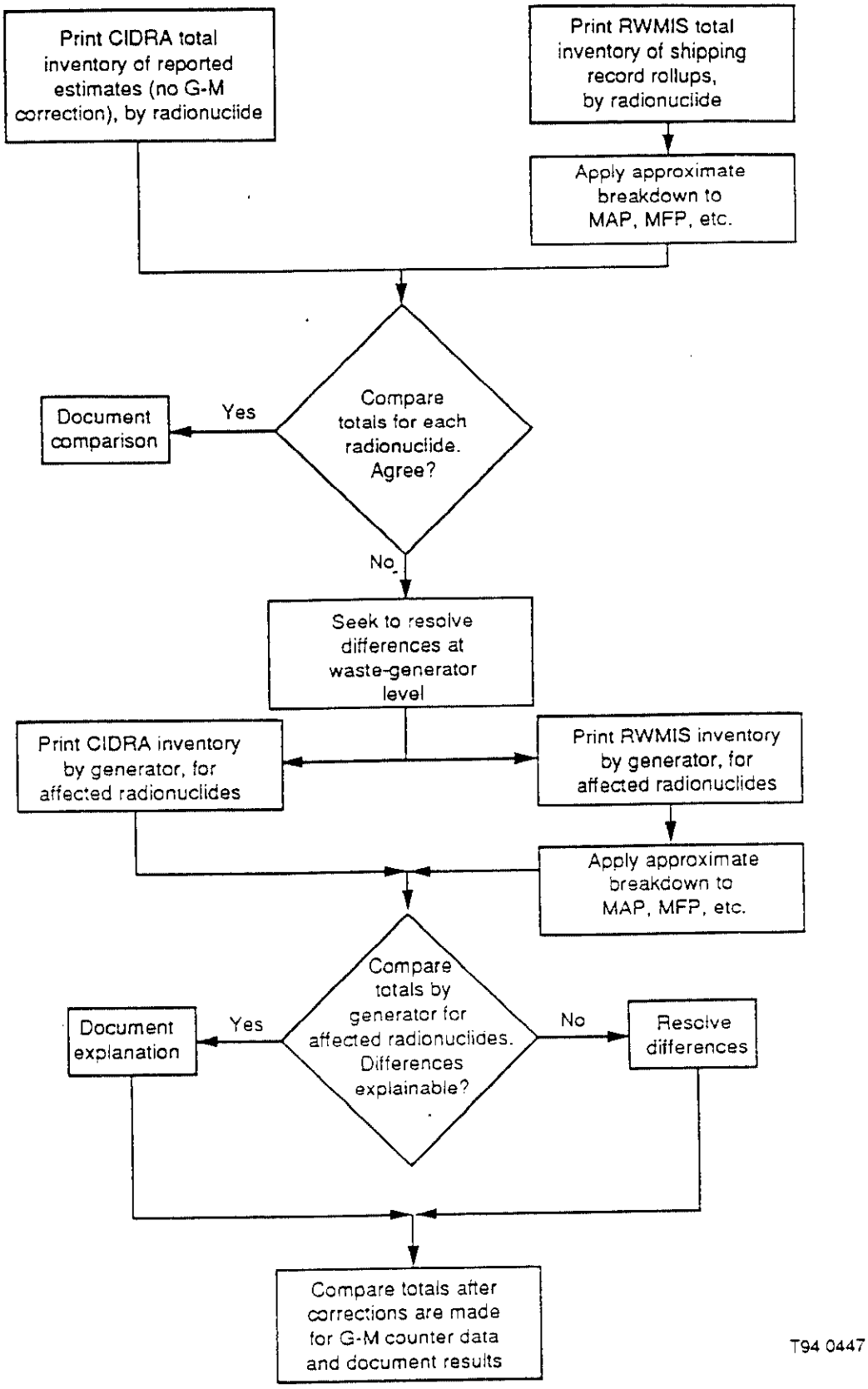

Figure 6-1. Approach for comparing the radionuclide inventory in CIDRA with that in the shipping record rollups of the RWMIS.

Table 6-3. Radionuclide inventories as given by RWMIS shipping record rollups and by CIDRA (with and without Geiger-Müller counter corrections): 1984–1993.

Radionuclide	RWMIS inventory (Ci)	RWMIS (with generic entries distributed) (Ci)	CIDRA reported estimates (no G-M corrections) (Ci)	CIDRA best estimate (with G-M corrections) (Ci)
Co-60	1.0E+06	1.0E+06	1.4E+06	1.4E+06
Ni-63	4.8E+05	4.8E+05	4.8E+05	4.8E+05
Co-58	4.5E+05	4.5E+05	2.0E+05	2.0E+05
Mn-54	3.2E+05	3.2E+05	1.2E+05	1.2E+05
H-3	3.0E+05	3.0E+05	3.0E+05	3.0E+05
Fe-55	1.5E+05	1.5E+05	1.6E+05	1.6E+05
Cr-51	3.7E+04	3.7E+04	4.7E+04	4.7E+04
Ta-182	2.0E+04	2.0E+04	1.8E+04	1.8E+04
Sn-119m	8.9E+03	8.9E+03	8.8E+03	8.8E+03
W-185	5.5E+03	5.5E+03	6.4E+03	6.4E+03
Cs-137	4.1E+03	6.4E+03	5.7E+03	3.1E+03
Nb-95	4.0E+03	4.0E+03	3.8E+03	3.8E+03
MFP	3.8E+03	0	0	0
Hf-181	3.8E+03	3.8E+03	3.4E+03	3.4E+03
Fe-59	3.8E+03	3.8E+03	1.5E+04	1.5E+04
Sb-125	2.9E+03	3.0E+03	2.9E+03	2.9E+03
MAP	2.9E+03	0	0	0
Hf-175	2.4E+03	2.4E+03	2.8E+03	2.8E+03
Zr-95	2.2E+03	2.2E+03	2.1E+03	2.1E+03
Ni-59	1.4E+03	1.4E+03	1.4E+03	1.4E+03
Zn-65	1.0E+03	1.0E+03	1.0E+03	1.0E+03
Ce-144	9.2E+02	1.6E+03	3.5E+02	2.1E+02
W-187	9.2E+02	9.2E+02	0	0
Sr-90	4.5E+02	9.8E+02	9.9E+02	5.8E+02
Cs-134	2.7E+02	2.9E+02	2.1E+02	1.4E+02
Pr-144	2.4E+02	2.7E+02	2.2E+02	1.1E+02
Unidentified beta-gamma	1.4E+02	0	0	0
Y-90	1.3E+02	1.4E+02	2.6E+02	2.0E+02
Ru-106	1.2E+02	1.3E+02	1.2E+02	6.4E+01
Rh-106	1.2E+02	1.3E+02	1.2E+02	6.1E+01
Sn-117M	1.2E+02	1.2E+02	1.2E+02	1.2E+02
Sc-46	5.0E+01	5.0E+01	5.0E+01	5.0E+01
Te-125m	4.2E+01	4.2E+01	4.2E+01	4.2E+01

Table 6-3. (continued).

Radionuclide	RWMIS inventory (Ci)	RWMIS (with generic entries distributed) (Ci)	CIDRA reported estimates (no G-M corrections) (Ci)	CIDRA best estimate (with G-M corrections) (Ci)
Sr-89	3.0E+01	3.0E+01	3.0E+00	3.0E+00
Sn-113	2.4E+01	2.4E+01	2.4E+01	2.4E+01
Eu-152	1.0E+01	1.0E+01	7.3E+00	4.1E+00
Th-228	1.0E+01	1.0E+01	1.0E+01	1.0E+01
C-14	1.0E+01	3.8E+01	7.1E+01	4.0E+01
Eu-154	6.0E+00	6.0E+00	6.0E+00	3.3E+00
Ba-137m	5.5E+00	5.5E+00	5.2E+00	4.6E+00
Ce-141	3.0E+00	3.0E+00	2.9E+00	2.9E+00
Na-24	2.7E+00	2.7E+00	2.7E+00	2.7E+00
La-140	2.6E+00	3.1E+00	3.1E+00	2.8E+00
Pm-147	2.4E+00	2.4E+00	2.4E+00	2.4E+00
U-238	2.3E+00	2.3E+00	1.6E+00	1.6E+00
U-232	2.2E+00	2.2E+00	2.2E+00	2.2E+00
Ba-140	2.2E+00	2.7E+00	2.7E+00	2.4E+00
Eu-155	2.1E+00	8.9E+01	6.6E+01	3.9E+01
Ag-110M	1.9E+00	1.9E+00	3.3E-02	1.8E-02
Mn-56	1.3E+00	1.3E+00	1.3E+00	1.3E+00
Gd-153	1.3E+00	1.3E+00	1.3E+00	1.3E+00
Ra-226	1.2E+00	1.2E+00	1.1E+00	1.1E+00
I-132	1.0E+00	1.0E+00	1.0E+00	1.0E+00
Ba-La-140	1.0E+00	0	0	0
Pu-239	3.0E-01	3.0E-01	2.6E+00	2.4+00
Pu-241	1.0E-01	1.0E-01	3.4E+01	1.7E+01
Am-241	3.9E-01	3.9E-01	4.0E+00	3.7E+00
U-234	1.5E-02	1.5E-02	3.5E+00	3.5E+00
Ag-110	7.5E-02	7.5E-02	1.9E+00	1.9E+00
Co-57	4.7E-02	4.7E-02	1.5E+00	1.5E+00
Tc-99	7.6E-03	2.5E+00	1.0E+00	5.0E-01
Nb-94	4.0E-02	4.0E-02	2.0E-01	2.0E-01
I-129	5.4E-04	6.8E-04	4.2E-03	2.1E-03
Total	2.8E+06	2.8E+06	2.8E+06	2.8E+06

Radionuclides were included in the comparison if their activity listed in RWMIS was at least 1 Ci. Additional radionuclides were included at the end of the list if their activity in the CIDRA database was at least 1 Ci before correction for the bias in the G-M counter readings. In addition, I-129 and Nb-94 were included because, although their activities were very small, they are very long-lived and relatively mobile when released from confinement.

In order to compare the CIDRA and RWMIS entries on the same basis, the generic terms had to be eliminated from the RWMIS entries. The activity represented by the generic terms was broken down as described in Section 6.2.2.1, leading to the values in the third column of Table 6-3. Also, dual radionuclide entries in RWMIS, such as Zr-Nb-95, were assigned as described in Appendix E. (Section 3.1.1 discusses the treatment of secular equilibrium in the CIDRA inventory and in the risk assessment.) The third column, therefore, represents the radionuclide inventory if RWMIS is used and the generic terms and dual radionuclide entries are broken down into their constituent radionuclides, as per the general methods used in the current study.

The fourth column gives the CIDRA values for the same radionuclides. The data in this column do not reflect the corrections made for the bias in inventory information based on the G-M counter surveys of waste containers. Thus, the data in this column are not the final CIDRA data, but are a version used only to check for completeness against the RWMIS values.

6.2.2.3 Comparisons of Results Before Applying Corrections to Activity Estimates Derived from G-M Counter Survey Data. The third and fourth columns of Table 6-3 enable comparisons of the results from CIDRA with those from RWMIS, with the generic radionuclide terms in RWMIS distributed using a simplified version of the CIDRA results, but without the effect of the corrections to data originally obtained from the G-M counter surveys. The following paragraphs discuss the results for only the predominant radionuclides. For both databases, data rollups by generator were consulted in evaluating the results, but generally are not presented here for brevity.

The nuclide-by-nuclide comparisons are discussed most easily by grouping the radionuclides according to fission products, activation products, and actinides. (Actinides include actinium and higher-numbered elements on the Periodic Table, such as plutonium, americium, and uranium.) Tritium (H-3) is a special case and is addressed first.

Tritium (H-3)—The CIDRA value agrees with the RWMIS entry to within the study precision of two significant figures. Virtually all of the H-3 is in stream TRA-670-1R, beryllium reflectors. Although the H-3 in this stream during the historical period (TRA-670-1H) was not reported in RWMIS, the H-3 in the stream was reported in RWMIS during the recent period. Hence, the agreement.

Fission Products—For the 8 fission products that constitute nearly all of this type of activity, the CIDRA and RWMIS values are compared below. They are discussed in the same order as their ranking in the CIDRA reported estimates.

The total activities of these 8 principal fission products in CIDRA (reported) and RWMIS (with generic entries distributed) differ by about 20%. The values are approximately 8,000 Ci and

10,000 Ci, respectively. The difference is less than the total random error for estimating the radioactivity in an individual waste shipment.

The distributions of the fission products in the two inventories also differ slightly. The sizes of the differences and the reasons for the differences are discussed below.

- **Cs-137.** The CIDRA value is approximately 11% smaller than the RWMIS value. The difference is less than the total random error for estimating the radioactivity in an individual waste shipment. Most of the Cs-137 is from TAN.

- **Sr-90.** The CIDRA value is approximately 1% larger than the RWMIS value. The difference is less than the total random error for estimating the radioactivity in an individual waste shipment. The Sr-90 is distributed fairly evenly over several generators.

- **Ce-144.** The CIDRA value is approximately one-fifth of the RWMIS value. Most of the Ce-144 is from ANL-W and from TRA in the MFP entries of RWMIS.

- **Y-90.** The CIDRA value is approximately twice the RWMIS value. As explained in Section 3.1, Y-90 is a short-lived decay product of Sr-90. Secular equilibrium is established quickly between the two radionuclides. Some preparers of waste information include the Y-90; some do not. The lack of full reporting of Y-90 is not important to the BRA; the calculations of radioactive decay performed in conjunction with the BRA will reflect equilibrium and the appropriate activity of Y-90.

- **Cs-134.** The CIDRA value is about three-fourths of the RWMIS value. CIDRA shows approximately 210 Ci, of which 92 Ci is from TAN, 66 Ci is from TRA, and 49 Ci is from PBF. RWMIS shows 290 Ci, of which 31 Ci is from TAN, 66 Ci is from TRA, 120 Ci is from ANL-W, 49 Ci is from PBF, and 20 Ci is from distributing generic entries. The difference is due primarily to a change in the distribution of radioactivity in the ANL-W waste.

- **Pr-144.** The CIDRA value is 18% smaller than the RWMIS value. The difference is less than the total random error for estimating the radioactivity in an individual waste shipment. Almost all of the Pr-144 is from ICPP.

- **Ru-106.** The CIDRA value is about 8% smaller than that of RWMIS. The difference is less than the total random error for estimating the radioactivity in an individual waste shipment. Almost all of the Ru-106 is from ICPP.

- **Rh-106.** The CIDRA value is about 8% smaller than that of RWMIS. The difference is less than the total random error for estimating the radioactivity in an individual waste shipment. Almost all of the Rh-106 is from ICPP.

I-129 is not one of the top 8 fission products in CIDRA in terms of activity. However, I-129 is important to the BRA because of its very long half-life (15.7 million years) and its potential for a comparatively high mobility in subsurface transport. The CIDRA value for I-129 is 4.2E-03 Ci,

almost entirely from TRA. The activity was estimated by means of the nuclear physics calculations described in Section 2.5.2. The RWMIS value is only 5.4E-04 Ci before distributing the generic entries, and 6.8E-04 Ci after. I-129 is seldom reported in waste shipments because it is very difficult to measure (EPRI 1987).

For the principal fission products and for the fission products as a whole, the comparison against the data in RWMIS confirmed that the CIDRA inventory of fission products is substantially complete. The only principal fission products for which the CIDRA values are substantially smaller than the RWMIS values are Cs-134 and Ce-144. These two radionuclides are relatively short-lived. The half-life of Cs-134 is only about 2 yr. The half-life of Ce-144 is less than 1 yr.

Activation Products—For the 18 activation products that constitute nearly all of this type of activity, the CIDRA and RWMIS values are compared below. They are discussed in the same order as their ranking in the reported estimates of CIDRA.

The total activities of these 18 principal activation products in CIDRA and RWMIS differ very slightly (2.47 million Ci and 2.49 million Ci, respectively).

The distributions of the activation products in the two inventories differ somewhat more. The sizes of the differences and the reasons for the differences are discussed below.

- **Co-60.** The CIDRA value is about 40% (400,000 Ci) larger than the RWMIS value. The difference arises primarily from the fact that the Co-60 for ANL-W waste is 370,000 Ci larger in CIDRA than in RWMIS. The ANL-W data reflect a more recent estimate of the distribution of activation products in the waste.

- **Ni-63.** The CIDRA value matches that of RWMIS. Nearly all of the Ni-63 is from NRF.

- **Co-58.** The CIDRA value is approximately one-half of the RWMIS value, for the same reason as discussed under Co-60. Nearly all of the Co-58 is from ANL-W.

- **Fe-55.** The CIDRA value is approximately 7% larger than the RWMIS value. The difference is less than the total random error for estimating the radioactivity in an individual waste shipment. Nearly all of the Fe-55 is from NRF.

- **Mn-54.** The CIDRA value is approximately one-third of the RWMIS value, for the same reason as discussed under Co-60. Nearly all of the Mn-54 is from ANL-W.

- **Cr-51.** The CIDRA value is approximately 25% larger than the RWMIS value, for the same reason as discussed under Co-60. Nearly all of the Cr-51 is from ANL-W.

- **Ta-182.** The CIDRA value is approximately 10% smaller than the RWMIS value. The difference is less than the total random error for estimating the radioactivity in an individual waste shipment. Nearly all of the Ta-182 is from NRF.

- **Fe-59.** The CIDRA value is approximately four times the RWMIS value. Nearly all of the Fe-59 is from ANL-W. The reason for the difference is discussed under Co-60.

- **Sn-119m.** The CIDRA value matches that of RWMIS within about 1%. Nearly all of the Sn-119m is from NRF.

- **W-185.** The CIDRA value is about 16% larger than the RWMIS value. The difference is less than the total random error for estimating the radioactivity in an individual waste shipment. All of the W-185 is from NRF.

- **Nb-95.** The CIDRA value is approximately 5% smaller than the RWMIS value. The difference is less than the total random error for estimating the radioactivity in an individual waste shipment. Almost all of the Nb-95 is from NRF.

- **Hf-181.** The CIDRA value is approximately 10% smaller than the RWMIS value. The difference is less than the total random error for estimating the radioactivity in an individual waste shipment. Almost all of the Hf-181 is from NRF.

- **Sb-125.** The CIDRA value is 3% smaller than that of RWMIS. The difference is less than the total random error for estimating the radioactivity in an individual waste shipment. Almost all of the Sb-125 is from NRF.

- **Hf-175.** The CIDRA value is approximately 17% larger than the RWMIS value. The difference is less than the total random error for estimating the radioactivity in an individual waste shipment. Almost all of the Hf-175 is from NRF.

- **Zr-95.** The CIDRA value is approximately 5% smaller than the RWMIS value. The difference is less than the total random error for estimating the radioactivity in an individual waste shipment. Almost all of the Zr-95 is from NRF.

- **Ni-59.** The CIDRA value matches that of RWMIS. Nearly all of the Ni-59 is from NRF.

- **Zn-65.** The CIDRA value matches that of RWMIS. All of the Zn-65 is from TRA.

- **W-187.** CIDRA shows no W-187 in the waste, whereas RWMIS shows 920 Ci, all from NRF. NRF personnel indicate that the W-187 entry in RWMIS is erroneous and should be W-185 instead. Making this correction would increase the RWMIS activity for W-185 such that it matches the CIDRA activity.

C-14, Tc-99, and Nb-94 are not among the top 18 activation products in CIDRA in terms of activity. However, they are important to the BRA because of their very long half-lives (5,730 years for C-14, 213,000 years for Tc-99, and 20,000 years for Nb-94) and their potential for comparatively high mobilities in subsurface transport. Their activities are discussed below.

The CIDRA value for C-14 is 71 Ci, of which 61 Ci is from TRA and 10 Ci from NRF. The RWMIS value (before distributing the generic entries) is only 10 Ci, consisting almost solely of the 10 Ci from NRF. The NRF value in both CIDRA and RWMIS was developed by means of nuclear physics calculations. The TRA value in CIDRA was developed by means of nuclear physics-based scaling factors. The TRA value in RWMIS is <1 Ci because C-14 is very difficult to measure in waste shipments; evidently, nuclear physics calculations were not performed to support the TRA data submittal to RWMIS.

The CIDRA value for Tc-99 is 1.0 Ci, almost all from TRA. RWMIS shows <0.01 Ci of Tc-99 (before distributing the generic entries). The explanation for the different inventories of this difficult-to-measure radionuclide is the same as that for C-14.

The CIDRA value for Nb-94 is 0.2 Ci, almost all from TRA. RWMIS shows 0.04 Ci. The explanation for the different inventories of this difficult-to-measure radionuclide is the same as that for C-14.

Among the principal activation products, the CIDRA inventory is substantially less than that in RWMIS only for Co-58 and Mn-54 (other than the erroneous RWMIS entry for W-187). Both of these radionuclides have half-lives of less than 1 year. Thus, for the principal activation products and for the activation products as a whole, the comparison against the data in RWMIS confirmed that the CIDRA inventory of activation products is substantially complete.

Actinides—The activity of actinides in the inventory for the recent period (1984–1993) is much smaller than that in the inventory for the historical period (1952–1983). The reason is that waste from RFP, which contained large quantities of actinides, was not disposed of at the SDA during the recent period.

For the eight actinides that constitute nearly all of this type of activity, the CIDRA and RWMIS values are compared below. The total activity of these eight principal actinides in CIDRA is much greater than the corresponding total in RWMIS (approximately 59 Ci versus 17 Ci). These total activities are not particularly large, however.

The sizes of the differences and the reasons for the differences are discussed below.

- **Pu-241.** The CIDRA value is 340 times the RWMIS value, although neither value is large (34 Ci versus 0.1 Ci). Nearly all of the Pu-241 in CIDRA is from TRA. The reason for the difference is that the Pu-241 from TRA was estimated using the nuclear physics calculation methods described in Section 2.5.2. The shipping records seldom report the Pu-241 that may be present in small activities, because of the difficulty in measuring it.

- **Th-228.** The CIDRA value matches that of RWMIS. Nearly all of the Th-228 is from ANL-E.

- **Am-241.** The CIDRA value is approximately 10 times the RWMIS value, although neither value is large (4 Ci versus 0.39 Ci). Nearly all of the Am-241 is from TRA. The reason for the difference is the same as that discussed under Pu-241.

- **U-234.** The CIDRA value is approximately 230 times the RWMIS value, although neither value is large (3.5 Ci versus 0.015 Ci). The U-234 in CIDRA is from several generators. The reason for the difference is that the RPDT data gatherers used nuclear physics calculations to ascribe U-234 to any substantial entries for U-235 and U-238. The amount of attributed U-234 depends on the quantity of uranium and the degree of enrichment, as described in Section 3.1. In the shipping records that provided the data for RWMIS, U-234 was seldom listed for shipments of uranium.

- **Pu-239.** The CIDRA value is approximately 9 times the RWMIS value, although neither value is large (2.6 Ci versus 0.3 Ci). Most of the Pu-239 in CIDRA is from ANL-W.

- **U-232.** The CIDRA value matches that of RWMIS. All of the U-232 is from ANL-E.

- **U-238.** The CIDRA value of 1.6 Ci is about 70% of the RWMIS value of 2.3 Ci. Both the CIDRA value and the RWMIS value include approximately 1.0 Ci from the SMC facility at TAN, the largest generator of the U-238.

  The next largest generator of the U-238 is ANL-E. This is the primary source of the different values for U-238. For ANL-E, RWMIS lists 0.75 Ci of U-238, as well as 0.00073 Ci of U-234 and 0.0048 Ci of U-235, for a total of 0.756 Ci of uranium. For ANL-E, CIDRA lists 0.11 Ci of U-238, as well as 0.71 Ci of U-234 and 0.030 Ci of U-235, for a total of 0.85 Ci of uranium. The RPDT data gatherer for the ANL-E waste reviewed the RWMIS printouts. It was concluded that the distribution of uranium radionuclides in RWMIS was not representative of the enrichment of uranium that would be expected in waste from ANL-E activities. The ratio of U-234 activity to U-238 activity in RWMIS was 0.00073/0.75 = 0.001 and did not correspond to any possible enrichment of uranium. [The values of the U-234/U-238 activity ratio should range from approximately 0.32 for depleted uranium to 100 or more for the most highly enriched uranium (Rich et al. 1988, EG&G Idaho 1985)]. The data gatherer determined to adjust the uranium radionuclide distribution. It was assumed that the uranium in the principal ANL-E waste stream resulted from research on the nuclear fuel for EBR-II, the ANL-W-operated reactor for which ANL-E provides technical support. A common enrichment for the uranium in EBR-II fuel is 50% U-235 by mass. For such an enrichment, the activity breakdown is approximately 95% U-234, 4% U-235, and 1% U-238 (Rich et al. 1988, EG&G Idaho 1985). Thus, an appropriate value for the activity ratio of U-234/U-238 would be in the neighborhood of 95, not 0.001. The RPDT data gatherer adjusted the relative activities of uranium radionuclides in the principal uranium-bearing stream, ANL-317-2R, in CIDRA, accordingly. The total amount of uranium was maintained essentially constant. (Other ANL-E streams contained smaller quantities of uranium and were not modified in the described manner, so the final CIDRA activity ratio does not equal 95.) The net result of this improved estimating process was to change much of the U-238 activity reported in RWMIS to U-234 activity, as entered in CIDRA. Thus, although the activity of U-238 from ANL-E in CIDRA is less than that in RWMIS, the activity of U-234 in CIDRA is correspondingly greater than that in RWMIS. (Also, see the discussion under U-234.)

- **Ra-226.** The CIDRA value is approximately 8% less than the RWMIS value. The difference is less than the total random error for estimating the radioactivity in an individual waste shipment. All of the Ra-226 is from ANL-E.

The only principal actinide in the CIDRA inventory whose activity is substantially less than that in RWMIS is U-238. That difference resulted from the improved method for estimating the distribution of the uranium radionuclides. Thus, the comparison against the data in RWMIS confirmed that the CIDRA inventory of actinides is substantially complete.

Total Inventory—The total activity in CIDRA (without the G-M corrections) is 2.8 million Ci; the total inventory in RWMIS is also 2.8 million Ci. The two totals agree to within the precision used in this study (two significant figures).

Conclusion—For the principal, longer-lived radionuclides (i.e., half-lives beyond 1 or 2 years) in each segment of the inventory—fission products, activation products, and actinides—the activity in CIDRA is similar to or larger than that in RWMIS. In addition, the total inventories over all radionuclides agree to within the precision used in this study. Therefore, the results of these comparisons of CIDRA values (without the G-M correction) against RWMIS values (with the generic activity terms distributed) confirm that the current task has not overlooked any substantial radioactivity in the waste.

6.2.2.4 Comparisons of Results After Applying CIDRA Corrections for G-M Counter Survey Data. The third and fifth columns of Table 6-3 enable comparisons of CIDRA and RWMIS results, including the effect of the corrected data from G-M counter surveys. Because of the corrections made to some of the values taken from the records, this comparison is less useful than the preceding one in identifying possible oversights in CIDRA. However, the comparison is useful to show the overall change in contaminant inventory. The following paragraphs discuss the impacts of the corrections, in reference to the comparisons against RWMIS.

The effect of the correction to the data derived from G-M counter surveys is to reduce the activities of certain radionuclides in the CIDRA inventory. This reduction arises in the following way. For individual waste streams, the reduction ranges from no change to a factor of 2. If the uncertainty in contaminant quantity was specified by the data gatherer, based on consideration of how the estimates or measurements were made originally, the G-M correction is not applied. If no uncertainty was specified (because the standard G-M counter method was believed to have been used), all activities in the waste stream were divided by two.

If all waste streams contributing to the inventory of a given radionuclide were subject to the factor of 2 reduction, then the total inventory of that radionuclide (last column of Table 6-3) reflects a reduction by a factor of 2, compared with the entry in the preceding column. For example, such is the case for Pr-144. On the other hand, if none of the contributory streams were subject to the correction, then the entries in the last two columns are identical. For example, the H-3 comes almost entirely from TRA waste streams, in which a nuclear physics-based calculational method was used rather than the G-M counter survey method. The entries for H-3 in the last two columns are, therefore, identical. For some radionuclides, the amount of the correction falls between these two extremes.

For radionuclides not affected by the G-M counter correction, such as H-3, the discussion in the previous comparison against RWMIS still applies. For radionuclides strongly affected by the correction, the CIDRA quantity is reduced by as much as a factor of 2, and the comparison against RWMIS is similarly affected. Only a few of the principal radionuclides are so affected.

Applying the G-M counter correction does not reduce the total activity in CIDRA substantially. To the precision of the two significant figures, the total is unchanged at 2.8 million Ci.

6.2.3 Comparisons at the Level of Individual Generators, Summed Over All Radionuclides

6.2.3.1 Approach. The RWMIS shipping record rollups were used for the comparisons at the level of individual generators. The methods used were basically the same as those described in Section 6.2.2. The principal difference is that the total radioactivity in the waste from each major generator in 1984–1993 is given.

Again, it is stressed that the comparisons presented here are for the purpose of confirming the general completeness of CIDRA. The comparisons are not intended to drive the totals from CIDRA to match those in RWMIS because CIDRA contains significantly improved information that is not found in RWMIS.

6.2.3.2 Comparisons. Table 6-4 provides the results of these comparisons. For confirming the completeness of CIDRA and for understanding the nature of the data-gathering process, the column containing the CIDRA reported estimates (no G-M correction) is compared with the RWMIS column to the left of it. The last column is shown only for perspective. The comparisons are discussed in terms of approximate numbers because of rounding all totals to two significant figures.

- **TAN.** The CIDRA value of 4,400 Ci for the total radioactivity in TAN waste is slightly larger than the RWMIS value of 3,700 Ci. The difference is due to (a) including about 300 Ci in waste from LOFT (later termed the Containment Test Facility) with TAN in CIDRA, but separately in RWMIS, and (b) the fact that CIDRA includes about 460 Ci of Sr-90 that was not reported in RWMIS because it is difficult to measure.

- **TRA.** The CIDRA value of 330,000 Ci for the total radioactivity in TRA waste is slightly larger than the RWMIS value of 320,000 Ci. The difference is probably due to the fact that some of the scaling factors used for TRA waste in the RPDT total more than unity (see Section 2.5.2). Such scaling factors include those for radionuclides whose radiation was not detected by the G-M counter.

- **ICPP.** The CIDRA value of 1,300 Ci is slightly smaller than the RWMIS value of 1,400 Ci. The difference of about 7% is considered to be within the uncertainty of the inventory approaches used.

Table 6-4. Radioactivity totals by generator as given by RWMIS shipping record rollups, and by CIDRA (with and without Geiger-Müller counter corrections): 1984–1993.

Major generator	RWMIS shipping record rollups (Ci)	CIDRA reported estimates (no G-M corrections) (Ci)	CIDRA best estimate (with G-M corrections) (Ci)
TAN	3.7E+03	4.4E+03	2.2E+03
TRA	3.2E+05	3.3E+05	3.2E+05
ICPP	1.4E+03	1.3E+03	6.7E+02
NRF	9.7E+05	9.7E+05	9.7E+05
ANL-W	1.5E+06	1.5E+06	1.5E+06
Other	3.9E+03	3.6E+03	3.2E+03
Total	2.8E+06	2.8E+06	2.8E+06

- **NRF.** The CIDRA value of 970,000 Ci matches the RWMIS value.
- **ANL-W.** The CIDRA value of 1.5 million Ci matches the RWMIS value.
- **Other.** The CIDRA value of 3,600 Ci for the total radioactivity in waste from the other generators is slightly smaller than the value of 3,900 Ci in RWMIS. The difference is explained by the fact that the RPDT included the approximately 300 Ci in the waste from LOFT with TAN, rather than with the other generators. This fact also helps explain why the CIDRA activity for TAN (including the 300 Ci from LOFT) is higher than that in RWMIS.

In summary, the generator-by-generator comparisons indicate that CIDRA values match the corresponding RWMIS values if the data uncertainties and the particular organization of the data by major generator are considered.

6.3 Comparison of the Inventory with Contaminants Detected in Environmental Monitoring

6.3.1 Purpose

It is useful to compare the estimated inventory of contaminants in CIDRA with the list of contaminants whose presence is detected at the RWMC by means of environmental monitoring. Potential gaps in the inventory may thereby be identified.

The following sections include (a) the approach used here to analyze contaminant monitoring results; (b) a summary of routine environmental monitoring activities and of special studies not part of the routine monitoring; (c) a brief summary of the monitoring results in terms of contaminants detected, years, and environmental media; and (d) comparisons of contaminants

detected against the contaminant inventory in CIDRA for the historical and recent periods. (Because the environmental monitoring may detect contaminants disposed of during either the historical or the recent period, the comparison was performed simultaneously for the inventory of both periods.) The documents from which the monitoring summaries were produced are listed in the bibliography in Appendix F.

6.3.2 Approach

Pertinent monitoring data for the RWMC were obtained from two primary sources: (a) annual summary reports for routine monitoring and (b) documentation for special environmental studies. Routine monitoring results for the Environmental Monitoring Program have been summarized annually since 1976. Concentrations are measured for radiological and nonradiological contaminants in air, soil, water, geologic media, and biotic media. These data were examined and summarized for the years 1976 through 1993. Existing databases and documents were consulted to identify special studies conducted on the SDA that resulted in reported environmental concentrations for radiological or nonradiological contaminants. Routine monitoring and special study results were evaluated by contaminant and medium and were summarized. The monitoring results were compared with the list of contaminants in the CIDRA inventory. The results of the comparison were interpreted with respect to the completeness of the list of contaminants in the inventory.

6.3.3 Environmental Monitoring Program

A comprehensive monitoring program is conducted at the RWMC and other areas of the INEL. The program provides for routine monitoring and data interpretation of radioactive and nonradioactive contaminants in the environment associated with the RWMC/SDA (Wilhelmsen et al. 1994).

Routine monitoring activities conducted as part of the program for the RWMC/SDA are summarized in Table 6-5. The program includes measurement of the concentrations of radioactive contaminants in air, water, soil, and biota (vegetation and small mammals), as well as monitoring of ambient radiation (Wilhelmsen et al. 1994). Monitoring conducted by RESL and groundwater monitoring activities conducted by the U.S. Geological Survey (USGS) are incorporated into the program and included in the annual summary reports. Nonradiological contaminants—metals and organics in liquid effluents and drinking water—are also assessed.

6.3.4 Special Studies

A number of special or one-time environmental studies for radiological and nonradiological contaminants have been performed at the RWMC/SDA. Data collected as part of the RWMC Subsurface Investigations Program, USGS studies, and other contaminant investigative studies were reviewed and summarized. Investigations included subsurface drilling, soil vapor monitoring, and groundwater monitoring. Data from the studies included in this task date back as far as the mid-1970s.

Table 6-5. Routine environmental monitoring activities performed at the Subsurface Disposal Area (compiled from Wilhelmsen et al. 1994).

Activity	Facility	Description	Frequency of analysis	Type of analysis
RADIOLOGICAL CONTAMINANTS				
Ambient air monitoring	SDA	Eight low-volume air samples operated at 0.14 m^3/min (includes one control and one replicate)	Semimonthly Semimonthly Monthly Quarterly	Gross alpha Gross beta Gamma spectrometry Radiochemistry[a]
Soil sampling	SDA	Five locations in each of five major areas (plus one control area)	Triennially	Gamma spectrometry Radiochemistry[a]
Subsurface water (sampled by the USGS)	SDA	2-L samples from each of six wells (five wells to the aquifer, one well to perched water)	65-m (perched water) well annually 183-m (aquifer) wells quarterly Production well quarterly	Gamma spectroscopy, chlorides (i.e., Cl-35), H-3, Sr-90, Co-60, Cs-137, Pu-238, Pu-239/240, and Am-241
Surface water sampling	SDA	4-L surface runoff samples from SDA and control location	Quarterly, but depends on precipitation	Gross alpha Gross beta Gamma spectrometry Radiochemistry[a,b,c]
Biotic surveillance	SDA and TSA	Small mammals—three composites in each of five major areas (plus one control area)[c]	Annually, but species sampled varies each year depending on availability	Gamma spectrometry Radiochemistry[a]
		Vegetation—three composites in each of five major areas (plus one control area)[c]	Annually, but species sampled varies each year depending on availability	Gamma spectrometry Radiochemistry[a]
		Small mammal burrow excavations (soil)—three composites from each of five major areas	Annually	Gamma spectrometry Radiochemistry[a]
NONRADIOLOGICAL CONTAMINANTS				
Subsurface water (sampled by the USGS)		Drinking water	Production well monthly	Organics Specific conductance Chloride, sodium, nitrate

a. Analysis for Am-241, Pu-238, Pu-239/240, U-235, U-238, and Sr-90.

b. Samples for radiochemical analyses usually taken during second quarter only.

c. Exact number of samples may vary due to availability.

6.3.5 Summary of Monitoring Results

The results of routine monitoring and special studies for radiological and nonradiological contaminants in the SDA are summarized in Appendix F.

6.3.6 Comparison of Contaminants Detected in Monitoring Activities Against Contaminants Identified in the Waste Inventory

Table 6-6 compares the results from environmental monitoring against the results of the inventory compilation for the historical and recent periods. The table lists the contaminants detected in routine monitoring or in special studies, the presence or nonpresence of each contaminant in the waste inventory, the media in which the contaminants were detected, the years in which they were detected, and brief conclusions concerning the comparisons (i.e., monitoring reliability and the qualitative amount of the contaminant in historical and recent periods). The table lists radiological contaminants first, followed by nonradiological contaminants.

6.3.6.1 Radiological Contaminants. No radiological contaminants that were reliably detected in the monitoring were missing from the waste inventory.

The following radiological contaminants were detected in reliable data from the monitoring and were identified in the waste inventory: Am-241, Co-60, Cs-134, Cs-137, H-3, Pu-238, Pu-239/240, Sb-125, Sr-90, U-234, U-235, and U-238.

As stated previously, contaminants detected in monitoring at the SDA might not have migrated from the buried waste. This could be the case, for example, with contaminants that are detected only in the aquifer. As another example, U-234, U-235, and U-238 are detected from time to time at the SDA. However, these radionuclides also occur naturally. Only a carefully constructed set of control samples will discriminate as to the likely origin of these three detected radionuclides, between the naturally occurring source and the source within the buried waste. It is beyond the scope of this document to provide definitive determinations on the source of the contaminants detected in the monitoring. The purpose of the present comparison is a simple check to help ensure that the inventory has not omitted any contaminants whose possible presence in the buried waste is manifest by environmental monitoring data.

The following radiological contaminants were detected only in the years before improved routine monitoring began, about 1984 (as discussed in Appendix F, these detections are questionable): Ac-228, Ag-110, Ba-140, Ce-141, Ce-144, Cr-51, Eu-155, Fe-59, Hf-181, Hg-203, I-131, Mn-54, Nb-95, Pb-212, Ru-103, Ru-106, Sb-124, Sc-46, Ta-182, U-237, Y-91, Zn-65, and Zr-95. There are no known, reliable monitoring data suggesting the migration of these contaminants at the SDA. This conclusion is not surprising because many of these contaminants have extremely low mobilities (being trapped in metal matrices), many have very short half-lives, and many are present in relatively small amounts.

The historical inventory contains a large activity of Pu-241, and this radionuclide is not monitored. The reason is that Pu-241, a beta-emitter, is less radiotoxic than the alpha-emitting

Table 6-6. Comparison of results of environmental monitoring against results of the inventory compilation.

Contaminant	Contaminant present in inventory—historical period 1952–1983?	Contaminant present in inventory—recent period 1984–1993?	Environmental media in which detected[a]	Years detected[b]	Conclusion[c,d]
RADIOLOGICAL					
Ac-228	No	No	Aquifer	1979	Not identified in waste for either period; monitoring detections not reliable
Ag-110	Yes	Yes	Air Surface water Soil	1980 1977 1979, 80	Minute quantities identified in waste for both time periods; monitoring detections not reliable
Am-241	Yes	Yes	Aquifer Surface water Subsurface soil Surficial sediment Soil Biota—vegetation Biota—soil Biota—tissue Air	1976, 81, 82, 84, 87 1977, 83–85, 90–93 1975–77, 85–88, 89 1989 1977–81, 84, 86, 88, 91, 92 1984, 86, 87, 90–93 1984–86, 90 1987, 89 1978–81, 84–93	Very large and minute quantities identified in waste for historical period and for recent period, respectively; detected frequently in monitoring program
Ba-140	Yes	Yes	Air	1980	Small and minute quantities identified in waste for historical period and for recent period, respectively; monitoring detections not reliable
Ce-141	Yes	Yes	Aquifer Surface water Soil Air	1983 1977, 81 1979–81 1978–81, 83–84	Small and minute quantities identified in waste for historical period and for recent period, respectively; monitoring detections not reliable
Ce-144	Yes	Yes	Aquifer Subsurface soil Surface water Soil Air	1976, 77, 79 1975–78 1976, 78, 79, 82 1978–81 1978–81, 83–84	Very large and moderate quantities identified in waste for historical period and for recent period, respectively; monitoring detections not reliable

Table 6-6. (continued).

Contaminant	Contaminant present in inventory—historical period 1952–1983?	Contaminant present in inventory—recent period 1984–1993?	Environmental media in which detected[a]	Years detected[b]	Conclusion[c,d]
Co-58	Yes	Yes	Soil	1978–81	Very large quantities identified in waste for both time periods; most monitoring detections not reliable
			Air	1978–81, 83, 85	
Co-60	Yes	Yes	Aquifer	1980, 87	Very large quantities identified in waste for both time periods; detected frequently in monitoring program
			Perched water	1976–77	
			Subsurface soil	1976–88, 89	
			Surface water	1977	
			Surficial sediment	1989	
			Soil	1977–81, 86	
			Biota—vegetation	1983	
			Biota—soil	1984	
			Biota—tissue	1987, 91	
			Air	1978–81, 83, 86	
Cr-51	Yes	Yes	Surface water	1977	Very large and large quantities identified in waste for historical and recent periods, respectively; monitoring detections not reliable
			Soil	1978–81	
			Air	1978–81, 83	
Cs-134	Yes	Yes	Surface water	1977, 79, 81	Moderate and small quantities identified in waste for historical and recent periods, respectively; detected occasionally in monitoring program
			Soil	1978–81	
			Biota—vegetation	1987	
			Air	1978–81, 85	
Cs-137	Yes	Yes	Aquifer	1976, 77, 80, 86, 87	Very large and moderate quantities identified in waste for historical and recent periods, respectively; detected frequently in monitoring program
			Perched water	1976, 77, 88	
			Subsurface soil	1975–88, 89	
			Surface water	1976, 77, 79–81, 83–86, 88, 90, 93	
			Surficial sediment	1989	
			Soil	1977–81, 84, 88, 89, 92	
			Biota—vegetation	1978, 83, 84, 87	
			Biota—soil	1984, 86, 90	
			Biota—tissue	1987, 91, 92	
			Air	1978–81, 84–85, 87, 91	

Table 6-6. (continued).

Contaminant	Contaminant present in inventory—historical period 1952–1983?	Contaminant present in inventory—recent period 1984–1993?	Environmental media in which detected[a]	Years detected[b]	Conclusion[c,d]
Eu-152	Yes	Yes	Surface water Soil Biota—tissue Air	1976, 78–79 1978–81 1987 1978–81	Small and minute quantities identified in waste for historical and recent periods, respectively; most monitoring detections not reliable
Eu-154	Yes	Yes	Subsurface soil Surface water Soil Biota—tissue Air	1985 1976, 79 1978–81, 89 1987 1978–81	Moderate and minute quantities identified in waste for historical and recent periods, respectively; most monitoring detections not reliable
Eu-155	Yes	Yes	Soil Air	1981 1981	Large and minute quantities identified in waste for historical and recent periods, respectively; monitoring detections not reliable
Fe-59	Yes	Yes	Aquifer Soil Air	1976 1979–81 1978–81	Large quantities identified in waste for both time periods; monitoring detections not reliable
H-3	Yes	Yes	Aquifer Perched water	1977–93 1976–77, 92, 93	Very large quantities identified in waste for both time periods; detected frequently in monitoring program
Hf-181	Yes	Yes	Soil Air	1978–81 1978–81	Minute and moderate quantities identified in waste for historical and recent periods, respectively; monitoring detections not reliable
Hg-203	Yes	No	Soil Air	1980–81 1978–81	Minute and no quantities identified in waste for historical period and recent period, respectively; monitoring detections not reliable
I-131	Yes	Yes	Air	1980	Minute quantities identified in waste for both time periods; monitoring detections not reliable

Table 6-6. (continued).

Contaminant	Contaminant present in inventory—historical period 1952–1983?	Contaminant present in inventory—recent period 1984–1993?	Environmental media in which detected[a]	Years detected[b]	Conclusion[c,d]
Mn-54	Yes	Yes	Soil Air	1979–81 1978–81, 83	Very large quantities identified in waste for both time periods; monitoring detections not reliable
Nb-95	Yes	Yes	Surface water Soil Air	1977 1978–81 1978–81	Moderate quantities identified in waste for both time periods; monitoring detections not reliable
Pb-212	Yes	No	Aquifer	1978	Minute and no quantities identified in waste for historical period and recent period, respectively; monitoring detections not reliable
Pu-238	Yes	Yes	Aquifer Perched water Subsurface soil Surface water Surficial sediment Soil Soil water Biota—vegetation Biota—tissue Air	1981, 83, 87 1976, 77, 89 1975, 89 1983 1989 1979–81, 88, 89, 91, 92 1989 1984, 86–87, 90 1987, 89 1980, 86–88	Large and small quantities identified in waste for historical period and recent period, respectively; detected frequently in monitoring program
Pu-239/240	Yes	Yes	Aquifer, perched Subsurface soil Surface water Surficial sediment Soil Soil water Biota—vegetation Biota—soil Biota-tissue Air	1976, 85–89 1975–78, 85–88, 89 1983–85 1989 1976–77, 79–81, 86, 88, 89, 91–93 1989 1986–87, 90 1984, 86–90 1987, 89 1980, 84–88, 90–93	Very large and small quantities identified in waste for historical period and recent period, respectively; detected frequently in monitoring program

Table 6-6. (continued).

Contaminant	Contaminant present in inventory—historical period 1952–1983?	Contaminant present in inventory—recent period 1984–1993?	Environmental media in which detected[a]	Years detected[b]	Conclusion[c,d]
Ru-103	Yes	Yes	Surface water Soil Air	1977, 81 1978–81 1978–80, 83	Small and minute quantities identified in waste for the historical and recent periods, respectively; monitoring detections not reliable
Ru-106	Yes	Yes	Surface water Soil Biota—vegetation Air	1976–77, 79 1979, 81 1978 1978–81	Moderate and small quantities identified in waste for the historical and recent periods, respectively; monitoring detections not reliable
Sb-124	Yes	Yes	Soil Air	1979–81 1979–81	Moderate and minute quantities identified in waste for the historical and recent periods, respectively; monitoring detections not reliable
Sb-125	Yes	Yes	Surface water Soil Biota—vegetation Biota—tissue Air	1978–81 1978–81 1987 1987 1978–81, 84	Very large and moderate quantities identified in waste for the historical and recent periods, respectively; detected occasionally in monitoring program, but early detections not reliable
Sc-46	Yes	Yes	Soil Air	1979–81 1978–81	Minute quantities identified in waste for both time periods; monitoring detections not reliable
Sr-90	Yes	Yes	Aquifer, perched Subsurface soil Surface water Surficial sediment Soil Biota—vegetation Biota—tissue Biota—soil Air	1976, 78–80, 85–88 1975–88, 89 1987 1989 1988, 91, 92 1983, 84, 86, 87, 90–93 1987, 89 1984 1986–88, 93	Very large and small quantities identified in waste for the historical and recent periods, respectively; detected frequently in monitoring program

Table 6-6. (continued).

Contaminant	Contaminant present in inventory—historical period 1952–1983?	Contaminant present in inventory—recent period 1984–1993?	Environmental media in which detected[a]	Years detected[b]	Conclusion[c,d]
Ta-182	Yes	Yes	Soil Air	1979–81 1979–81	Minute and large quantities identified in waste for the historical and recent periods, respectively; monitoring detections not reliable
U-234	Yes	Yes	Soil Biota—vegetation Biota—tissue	1986, 92 1985, 87 1987	Small quantities identified in waste for both time periods; detected occasionally in monitoring program
U-235	Yes	Yes	Soil Biota—vegetation Biota—tissue	1983 1987 1987, 89	Small quantities identified in waste for both time periods; detected rarely in monitoring program
U-237	No	No	Air	1980	Not identified in waste for either time period; monitoring detections not reliable
U-238	Yes	Yes	Soil Biota—vegetation Biota—tissue	1983–84, 92 1987 1987, 89	Small quantities identified in waste for both time periods; detected rarely in monitoring program
Y-91	Yes	No	Soil Air	1979–80 1979–80	Minute and no quantities identified in waste for historical period and recent period, respectively; monitoring detections not reliable
Zn-65	Yes	Yes	Soil Air	1979–81 1978–81	Small and moderate quantities identified in waste for the historical and recent periods, respectively; monitoring detections not reliable
Zr-95	Yes	Yes	Soil Air Surface water	1979–81 1978–81 1977	Large and moderate quantities identified in waste for the historical and recent periods, respectively; monitoring detections not reliable

Table 6-6. (continued).

Contaminant	Contaminant present in inventory—historical period 1952–1983?	Contaminant present in inventory—recent period 1984–1993?	Environmental media in which detected[a]	Years detected[b]	Conclusion[c,d]
NONRADIOLOGICAL					
Organics[e]					
1,1,1-trichloroethane	Yes	No	Aquifer, perched Soil borehole (vapor) Soil/soil gas Air	1987–93 1987, 88 1987 1991, 94	Very large quantity identified in waste for the historical period; detected frequently in monitoring program
1,1,2-trichloro-trifluoroethane	Yes	No	Perched water Borehole (vapor) Soil/soil gas Air	1987–90 1987–88 1987 1989	Large quantity in waste for the historical period; detected frequently in monitoring program
1,1-dichloroethylene	No	No	Aquifer, perched	1987–93	Not specifically identified in inventory; detected frequently in monitoring program
1,1-dichloroethane	No	No	Aquifer, perched	1987–93	Not specifically identified in inventory; detected frequently in monitoring program
Acetone	Yes	No	Sedimentary interbed Air	1987 1994	Large quantity identified in waste for the historical period; detected rarely in monitoring program
Carbon tetrachloride	Yes	No	Aquifer, perched Borehole (vapor) Soil/soil gas Air	1987–93 1987–88 1987, 92 1987, 89	Very large quantity identified in waste for the historical period; detected frequently in monitoring program
Chloroform	Yes	No	Aquifer, perched Borehole (vapor) Sedimentary interbed Air	1987–93 1987–88, 92 1987 1989, 94	Probable very large quantity identified in waste for the historical period; detected frequently in monitoring program

Table 6-6. (continued).

Contaminant	Contaminant present in inventory—historical period 1952–1983?	Contaminant present in inventory—recent period 1984–1993?	Environmental media in which detected[a]	Years detected[b]	Conclusion[c,d]
Organics (continued)					
Dichlorodifluoromethane	No	No	Aquifer, perched Air	1987–93 1994	Not specifically identified in inventory; detected frequently in monitoring program
Methylene chloride	Yes	No	Sedimentary interbed Perched water Air	1987 1993 1991, 94	Very large quantity in waste for the historical period; detected rarely in monitoring program
Phenol	No	No	Aquifer	1991	Not specifically identified in inventory; detected rarely in monitoring program
Tetrachloroethylene	Yes	No	Aquifer, perched Borehole (vapor) Soil/soil vapor Air	1987–93 1987, 92 1987 1994	Very large quantity identified in waste for the historical period; detected frequently in monitoring program
Toluene	Yes	No	Aquifer, perched Borehole (vapor) Air	1987–93 1987, 92 1994	Large quantity identified in waste for the historical period; detected frequently in monitoring program
Trichloroethylene	Yes	No	Air Aquifer, perched Soil borehole (vapor) Sedimentary interbed	1987, 89 1987–93 1987, 92 1987	Very large quantity identified in historical waste; detected frequently in monitoring program
Metals					
Antimony	Yes	No	Perched	1988, 93	Small quantity identified in waste for the historical period; detected rarely in monitoring program
Arsenic	No	Yes	Aquifer, perched	1987–88, 93	Only minute quantity identified in waste for recent period; detected rarely in monitoring program

Table 6-6. (continued).

Contaminant	Contaminant present in inventory—historical period 1952–1983?	Contaminant present in inventory—recent period 1984–1993?	Environmental media in which detected[a]	Years detected[b]	Conclusion[c,d]
Metals (continued)					
Barium	No	No	Perched water Sedimentary interbed	1988, 93 1987	Not identified in waste; detected rarely in monitoring program
Beryllium	Yes	Yes	Perched water Subsurface soil Sedimentary interbed	1988, 93 1991 1987	Very large and large quantities in inventory for the historical and recent periods, respectively; detected occasionally
Boron	No	No	Soil	1982	Not identified in inventory; not on lists of hazardous substances; detected rarely
Cadmium	Yes	Yes	Perched water Soil	1988, 93 1982	Large and small quantities of waste identified in historical and recent period, respectively; detected rarely
Chromium	Yes	Yes	Surface water Aquifer, perched Soil Sedimentary interbed	1986 1985–88, 93 1982 1987	Moderate and small quantities of waste identified in historical and recent period, respectively; detected occasionally
Cobalt	No	No	Perched water	1988, 93	Not identified as a nonradiological contaminant in waste; detected rarely in monitoring program
Copper	Yes	Yes	Perched water Soil Sedimentary interbed	1988, 93 1982 1987	Small and moderate quantities of waste identified in historical and recent period, respectively; detected occasionally
Lead	Yes	Yes	Perched water Soil	1988, 93 1982	Very large quantities identified in inventory for both periods; detected rarely
Mercury	Yes	Yes	Perched water Subsurface soil[f] Sedimentary interbed Soil vapor	1988, 93 1991[f] 1987 1990	Large and small quantities of waste identified in historical and recent period, respectively; detected occasionally in environmental monitoring; detected in direct sampling of the Acid Pit

Table 6-6. (continued).

Contaminant	Contaminant present in inventory—historical period 1952–1983?	Contaminant present in inventory—recent period 1984–1993?	Environmental media in which detected[a]	Years detected[b]	Conclusion[c,d]
Metals (continued)					
Nickel	Yes	No	Perched water Sedimentary interbed	1988, 93 1987	Moderate quantity identified in waste for historical period; detected rarely in monitoring program
Selenium	No	No	Sedimentary interbed Subsurface water, perched	1987 1987, 88 1993	Not identified in waste; detected rarely in monitoring program
Silver	Yes	No	Perched water Sedimentary interbed	1988, 93 1987	Moderate quantity identified in waste for historical period; detected rarely in monitoring program
Thallium	No	No	Perched water Sedimentary interbed	1988, 93 1987	Not identified in waste; detected rarely in monitoring program
Tin	No	No	Perched water Sedimentary interbed	1988 1987	Not identified as a nonradiological contaminant in waste; detected rarely in monitoring program
Vanadium	No	No	Perched water Sedimentary interbed	1988, 93 1987	Not identified in waste; detected rarely in monitoring program
Zinc	No	No	Perched water Soil Sedimentary interbed	1988, 93 1982 1987	Identified in waste inventory only in radioactive form (< 1 g mass); detected rarely
Other					
Chloride	Yes	No	Aquifer, perched Soil	1979, 82–93 1982	Very large quantity identified in waste for historical period; detected frequently in monitoring program
Cyanide	Yes	No	Perched water Sedimentary interbed	1988 1987	Small quantity identified in waste for historical period; detected rarely in monitoring program

Table 6-6. (continued).

Contaminant	Contaminant present in inventory—historical period 1952–1983?	Contaminant present in inventory—recent period 1984–1993?	Environmental media in which detected[a]	Years detected[b]	Conclusion[c,d]
Other (continued)					
Nitrate	Yes	No	Aquifer, perched Surface water Soil	1982–83, 85, 87, 93 1980–82 1980–83	Very large quantity identified in waste for historical period; detected frequently in monitoring program
Sodium ion	Yes	No	Aquifer, perched Surface water	1979, 82–93 1983–86	Very large quantity identified in waste for historical period; detected frequently in monitoring program
Sulfate	Yes	No	Perched water	1985, 88, 93	Very large quantity identified in waste for historical period; detected occasionally in monitoring program
Sulfide	No	No	Perched water Sedimentary interbed	1988 1987	Not identified in waste; detected rarely in monitoring program

a. Subsurface water includes samples from five wells that sample aquifer water and from one well that samples perched water. Separate entries are indicated where possible.

b. Data obtained by EG&G Idaho, Inc. routine monitoring before approximately 1983 are considered to be of lower reliability because, in many cases, no control samples were collected or the control samples were from inappropriate locations. In many cases, contaminants detected in these early samples may have originated in the airborne or waterborne emissions from other INEL facilities, rather than from the SDA.

c. The following scheme was used to express quantitative inventory values using a set of qualitative terms. The expression E+03, for example, covers entries between E+03 and 9.9E+03. Radiological contaminants—Alpha-emitters: Very large=E+04 and greater, Large=E+03, Moderate=E+02, Small=E+01 and less; Beta/gamma-emitters: Very large=E+05 and greater, Large=E+04, Moderate=E+03, Small=E+02, Minute=E+01 and less. Nonradiological contaminants—Very large=E+07 and greater; Large=E+05, E+06; Moderate=E+03, E+04; and Small=E+02 and less.

d. The frequency of detection is expressed in a qualitative hierarchy of terms: frequently, occasionally, rarely. The determination of the appropriate term is based on technical judgment after considering (a) the number of years in which the contaminant was detected, and (b) the number of years in which the contaminant was monitored.

e. During 1984–1993, no organic contaminants were disposed of in the SDA.

f. It is also noted that mercury was sampled for in one SDA trench in 1990 but was not detected. Mercury was sampled for and detected in the Acid Pit.

plutonium and americium radionuclides that are monitored (Pu-238, Pu-239/240, and Am-241). Pu-241 is more difficult to measure and is also much shorter-lived than the other radionuclides mentioned.

6.3.6.2 Nonradiological Contaminants. Routine monitoring for nonradiological contaminants at the RWMC began in the mid- to late 1980s. All of the data for nonradiological contaminants are considered sufficiently reliable for use in these comparisons.

Ten of the 13 organic contaminants that were detected in the monitoring are listed in the historical inventory. Those not specifically listed in the inventory are 1,1-dichloroethylene, 1,1-dichloroethane, and dichlorodifluoromethane. The detections were in both aquifer water and perched water. The detections were frequent. Any contaminants detected only in the aquifer could have originated at other upgradient INEL facilities. However, any contaminants detected in perched water could have originated in the buried waste.

Several possible explanations exist as to why some of the organic contaminants were detected in the monitoring but not identified specifically in either this inventory or other inventory reports. First, the waste information on which the inventory is based could simply be incomplete. Second, the contaminants could have been secondary species in a waste stream, wherein only the primary species were identified. Third, the contaminants detected in the monitoring could be degradation products originating from a contaminant that is listed in the inventory. These three organics are very similar in molecular structure to organic compounds that have been identified in the inventory in large quantities; 1,1-dichloroethylene is similar to trichloroethylene, 1,1-dichloroethane is similar to 1,1,1-trichloroethane, and dichlorodifluoromethane is similar to 1,1,2-trichloro-1,2,2-trifluoroethane. Therefore, there is a strong possibility that these are impurities or degradation products of substances that are listed in the inventory. It is beyond the scope of this comparison to distinguish definitively among these possible explanations for the fact that three organics were detected in the monitoring but not identified specifically in the inventory. The conclusion is that nearly all of the organic contaminants detected in the monitoring were identified in the inventory for the historical period.

Among the metals, only beryllium, chromium, copper, and mercury have been detected more than rarely in the monitoring (i.e., more than once or twice). All of these metals were identified in the inventory, in quantities ranging from small to very large for both the historical and recent periods. Several other metals have been detected once or twice in the monitoring: cadmium, lead, zinc, antimony, arsenic, cobalt, barium, nickel, selenium, silver, thallium, tin, boron, and vanadium. The measured concentrations approximate natural background levels in many cases. Some of these metals have been identified in the inventory for both the historical and recent periods. The conclusion is that the entire inventory includes all toxic metals that have been detected in the environment on more than rare occasions and at concentrations well above natural background.

The last class of nonradiological contaminants monitored is certain inorganic species. Sodium ion, chlorides, sulfates, and nitrates are detected occasionally to frequently by monitoring; they are listed in the inventory for the historical period in various forms and in very large quantities. Sulfides were detected once in the monitoring, but were not identified in the inventory for either time period. Again, these detected contaminants could have originated from naturally occurring

sources or from the waste. Cyanide has been detected on two occasions and is identified in the inventory for the historical period in a small quantity.

6.3.6.3 Conclusions. No radiological contaminants that were reliably detected in the monitoring are missing from the waste inventory.

For the nonradiological contaminants, other than rare detections or detections at concentrations near natural background levels, no metals or other inorganics on the list of hazardous substances were detected in the environmental monitoring but not listed in the inventory for one of the two time periods. Ten of the 13 organic contaminants that were detected in the monitoring are listed in the inventory for the historical period. The other three organic contaminants may be degradation products or impurities of contaminants that were identified in the inventory for the historical period, or may have originated from other INEL sources.

6.4 Contaminant Profile Data Sheets

The inventory is also presented in a simple yet informative form, called contaminant profile data sheets, see Appendix G. The data sheets provide a quick reference summary for most of the principal contaminants. Data sheets were prepared for contaminants that were among those present in the largest quantities. Appendix G of this report contains only the data sheet for C-14. All other data sheets are presented in the HDT report (LITCO 1995).

Each data sheet briefly lists typical physical and chemical forms and properties of the contaminant, common uses, general presence in the environment, toxicology, and the results of environmental monitoring at the SDA. For radiological contaminants, the radiological properties and radiotoxicity are also included.

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7. OBSERVATIONS AND CONCLUSIONS

The observations and conclusions for the contaminants in the recent (1984–1993) and projected (1994–2003) waste are as follows:

- The combined use of many types of information sources—process knowledge, operating records, nuclear physics calculations, reports, interviews, shipping records, the RWMIS database, waste generator forecasts, and others—was essential to achieve the present degree of completeness of the recent and projected inventories.

- For nonradiological contaminants, the inventory information that could be located or deduced for the 1984–1993 and 1994–2003 periods from information sources and that is compiled in CIDRA is believed to be substantially complete.

- The number and quantities of nonradiological contaminants identified in or projected to be in the waste being disposed of in the SDA decreased dramatically after 1984. For most (but not all) of those nonradiological contaminants, the presence of the contaminant could cause the waste to be designated as hazardous per RCRA. Beginning in 1984, DOE was required to come into compliance with RCRA, so acceptance of mixed waste for disposal at the SDA was discontinued in April 1984. An exception was made for contaminated lead used as radiation shielding, which was allowed for disposal as late as December 31, 1987.

- For the radiological contaminants, the inventory information that could be located or deduced for the 1984–1993 and 1994–2003 periods from information sources and that is compiled in the new CIDRA database is believed to be substantially complete.

- A considerable effort was devoted to estimating the uncertainty in the quantities of nonradiological and radiological contaminants. For the projected waste, this effort included estimating the uncertainty in waste generator forecasts.

- A considerable effort was devoted to breaking down the generic radioactivity terms MAP, MFP, and unidentified beta/gamma-emitters for each generator so that a specific distribution of radionuclides would be available for the risk assessment.

- The predominant (by mass) nonradiological contaminants identified in the waste were as follows: for the 1984–1993 period—lead, beryllium, asbestos, copper, zirconium oxide, and chromium; for the 1994–2003 period—beryllium, asbestos, and chromium.

- The predominant (by radioactivity at the time of disposal) radiological contaminants identified in the waste were as follows: for the 1984–1993 period—Co-60, Ni-63, H-3, Co-58, Fe-55, Mn-54, Cr-51, Ta-182, and Fe-59; for the 1994–2003 period—H-3, Co-60, Co-58, Ni-63, Mn-54, Fe-55, and Cr-51.

- To confirm its substantial completeness, the compiled recent (1984–1993) inventory of radiological contaminants was compared against the corresponding inventory in the RWMIS database. The results of these comparisons confirm that the current task has

not overlooked any substantial radioactivity in the waste. The total activity in CIDRA (without the G-M correction) agrees with the total inventory in RWMIS to within the accuracy allowed by the use of two significant figures. For all of the principal, longer-lived radionuclides, the activity in CIDRA is similar to or larger than that in RWMIS.

- The total activities of the fission products differ between CIDRA and RWMIS by about 20%. This difference is less than the total random error for estimating the radioactivity in an individual waste shipment.

- The CIDRA value agrees with the RWMIS entry for tritium (H-3) to within the study precision of two significant figures. The H-3 is almost entirely in the beryllium reflectors from the TRA waste.

- The total activities of the activation products differ <1% between CIDRA and RWMIS.

- The activities of C-14, Tc-99, and I-129 in CIDRA are considerably larger than those in RWMIS. These radionuclides are important because of their very long half-lives and their relatively high mobility if released from the waste form. These radionuclides are very difficult to measure in waste shipments. The values in CIDRA were developed by means of nuclear physics calculations.

- As an additional confirmation of its completeness, the compiled nonradiological inventory for the 1984–1993 period was compared against the information found in previous reports. Very few studies have been performed on the nonradiological contaminants buried in the SDA in the recent period. Therefore, the comparisons were of limited value but identified no evidence that the new inventory was incomplete.

- As an additional confirmation of its completeness, the compiled radiological inventory for the 1984–1993 period was compared against the information found in previous reports. Only one report contained data for the recent period. Because the data were nearly identical to those in the RWMIS database, no detailed comparison was carried out.

- The compiled radiological inventory for the projected period (1994–2003) was compared to the waste generator forecasts. Because the waste generator forecasts were the starting point for evaluating the projected waste, the close agreement with reported estimates in CIDRA is not surprising. As expected and consistent with the assumptions, the best-estimate CIDRA values (after the bias corrections and other refinements are applied) differ substantially from the generator forecasts.

- As a final confirmation of its substantial completeness, the recent inventory (1984–1993) of contaminants was compared against the list of contaminants detected in environmental monitoring at the RWMC. The historical inventory (1952–1983) was also included in the comparison. No radiological contaminants were reliably detected in the monitoring that had not been identified in either the historical or the recent inventories. The only nonradiological contaminants detected more than rarely in the

environmental monitoring that were not identified in the inventories were three volatile organic compounds: 1,1-dichloroethylene, 1,1-dichloroethane, and dichlorodifluoromethane. These three contaminants may be degradation products or impurities associated with closely related contaminants that were identified in the historical inventory. Detected contaminants also could have originated from sources other than the subject waste (e.g., in effluents from other INEL facilities or from other waste at the RWMC).

- A large quantity of information was assembled and entered into CIDRA on the physical and chemical forms of the waste streams and of the contaminants, and on the packaging of the waste streams.

- Even though the information now residing in CIDRA has been through multiple checks and reviews, the possibility exists for oversights and discrepancies. In addition, new information about the waste is identified from time to time in other INEL programs. As new information is discovered, the database will be revised as necessary.

References for Section 7

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